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Long Term Recharge of Trickling Filter Effluent into Sand

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Long Term Recharge of Trickling Filter Effluent Into Sand

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The rapid infiltration of trickling filter effluent onto natural delta sand beds at the Lake George Village Sewage Treatment Plant has been shown to produce the equivalent of tertiary treatment to the domestic wastewater since 1939 with no indication of exhaustion of the purification capacity. Most of the purification took place in the top 10 m of the sand. BOD, COD, alkylbenzenesulfonates, total and fecal coliforms, and phosphates were essentially completely removed in the sand system. Ammonia and organic nitrogen were converted to nitrates, some of which were removed under reducing conditions. Vertical velocities in the sand were measured between 0.6 m/day and 2.5 m/day. Horizontal velocities varied between 3 m/day and 12 m/day. Allowing weeds to grow on the sand beds increased the infiltration rate when the depth of liquid on the bed exceeded 0.3 m, but decreased the infiltration rate at shallower depths. A rapid sand infiltration system is recommended as a suitable means of providing tertiary treatment to domestic wastewater.

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LONG TERM RECHARGE OF TRICKLING FILTER EFFLUENT INTO SAND

by

Donald B. Aulenbach
Department of Chemical and Environmental Engineering
Rensselaer Polytechnic Institute
Troy, New York 12181

Grant No. R-803452

Project Officer

Lowell E. Leach
Wastewater Management Branch
Robert S. Kerr Environmental Research Laboratory
Ada, Oklahoma 74820

ROBERT S. KERR ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U. S. ENVIRONMENTAL PROTECTION AGENCY
ADA, OKLAHOMA 74820

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FOREWORD

The Environmental Protection Agency was established to coordinate administration of the major Federal programs designed to protect the quality of our environment.

An important part of the agency's effort involves the search for information about environmental problems, management techniques, and new technologies through which optimum use of the nation's land and water resources can be assured and the threat pollution poses to the welfare of the American people can be minimized.

EPA's Office of Research and Development conducts this search through a nationwide network of research facilities.

As one of these facilities, the Robert S. Kerr Environmental Research Laboratory is responsible for the management of programs to: (a) investigate the nature, transport, fate and management of pollutants in groundwater; (b) develop and demonstrate methods for treating wastewaters with soil and other natural systems; (c) develop and demonstrate pollution control technologies for irrigation return flows; (d) develop and demonstrate pollution control technologies to prevent, control or abate pollution from the petroleum refining and petrochemical industries; and (f) develop and demonstrate technologies to manage pollution resulting from combinations of industrial wastewaters or industrial/municipal wastewaters.

This report contributes to the knowledge essential if the EPA is to meet the requirements of environmental laws that it establish and enforce pollution control standards which are reasonable, cost effective and provide adequate protection for the American public.

William C. Galegar

William C. Galegar
Director
Robert S. Kerr Environmental Research Laboratory

PREFACE

The area around Lake George was concerned with potential pollution of the lake long before the passage of PL 92-500. Pollution control for the most populated area around Lake George Village was accomplished by the installation of a rapid infiltration system for the equivalent of tertiary treatment of the secondary treated domestic wastes. This system has been in operation continuously since 1939 and provided the facilities for a closer evaluation of the effectiveness of the sand infiltration technique for final treatment of domestic wastewaters.

By providing detailed information relating to the purification of wastewater in the soil, a decision can be made whether or not to install similar treatment systems at other locations. The information provided should aid in the design of similar treatment systems. Some of the major advantages of a land disposal system include the savings in chemicals, the lack of additional sludges to dispose of, resistance to shock loading, reduced cost where land is available at a reasonable price, savings in energy where extreme pumping is not required, and complete treatment including removal of phosphorus and conversion of nitrogen compounds to nitrates which may also be removed by appropriate treatment procedures. Thus, land application of wastewater must be considered as a viable means of domestic wastewater treatment and disposal.

ABSTRACT

The Lake George Village Sewage Treatment Plant has been providing the equivalent of tertiary treatment of domestic wastewater by applying wastewater which has been subjected to trickling filter purification and secondary sedimentation to a natural delta sand deposit by the rapid infiltration method. The sand system has been shown to remove decomposable substances such as 5-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), alkylbenzensulfates, coliforms and fecal coli, and ammonia and organic nitrogen. Soluble inorganic materials such as sodium, chloride, potassium, and nitrate generally passed through the bed unchanged. Calcium and magnesium showed no significant changes; whereas alkalinity increased due to carbonate and bicarbonate reactions within the soil system. Orthophosphates were completely removed, and nitrates were somewhat removed under reducing conditions produced in some of the deeper sand beds.

Operation has been continued successfully throughout the cold winter months experienced in this area. Allowing grass to grow in the infiltration beds provided an increased infiltration rate at flooding depths exceeding 0.3 meters (m) (1 foot, ft); whereas a decrease in infiltration rate was observed under shallower depths of water. The greatest removal of constituents occurred in the top 10 m (30 ft) of most of the sand beds; however, continued quality improvement was observed through further vertical flow and approximately 600 m (200 ft) of horizontal flow. There does not seem to be any loss of the capacity of the complete system to remove the nutrients.

Both vertical and horizontal flow studies were made. The average vertical velocity in bed N-11 was 0.63 m/day (2 ft/day). The horizontal velocity varied throughout the area but was in a general range of 3 to 12 m/day (1 to 4 ft/day).

Studies at the Lake George Village Sewage Treatment Plant have shown that a rapid infiltration system for purifying secondary treated effluent can provide the equivalent of tertiary treatment with nutrient removal for an extended period of time exceeding at least 38 years. This information should be useful in providing data for the design of similar systems serving other areas of small population.

This report is submitted in fulfillment of Grant #R803452 by Rensselaer Polytechnic Institute (RPI) under the partial sponsorship of the U. S. Environmental Protection Agency. This report covers the period from November 1974 through June 30, 1977, but also includes data collected by the New York State Department of Environmental Conservation (NYSDEC) between June 1973 and October 1974, and work was completed as of November 17, 1978.

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The installation of the initial wells was conducted by Dr. T. James Tofflemire of the New York State Department of Environmental Conservation (NYSDEC). Some initial quality measurements were also conducted by NYSDEC.

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SECTION 1

INTRODUCTION

LAND TREATMENT OF WASTEWATER

Today's interest in land treatment of wastewater stems from man's increasing concern for improving the quality of the environment. Public Law 92-500 has influenced the development of improved waste treatment methods within the framework of increased efficiency, greater energy conservation, and a general reduction of operating costs. The potential water reuse factor achieved with land treatment has received particular attention. The U. S. Environmental Protection Agency (EPA) has supported research and promoted its public use.

Concern for zero pollution discharge under Public Law 92-500 has rekindled interest in land treatment of wastewater. Land treatment is intimately linked with ground water. Analysis of the existing and expected ground water quality is vital in the implementation of a land treatment system. There is justifiable concern with pollution of the ground water, capacity of the system (rising ground water tables are very troublesome), and the ultimate residence of the water. The most exciting parts of land treatment are the recycle aspects. Under ideal conditions both nutrient recycle and water recycle are possible. Storage of water underground is desirable due to minimal evaporative losses, cool temperatures, slow flow rates, and the natural filtering capacity of the soil.

Specifically, this report will concern itself with the rapid infiltration form of land treatment which is one of the three basic forms of land treatment. These forms include: spray irrigation (SI), overland flow (OF), and rapid infiltration (RI). Rapid infiltration, or infiltration-percolation as it is also referred to, (Figure 1) involves: (1) the flooding of a basin with between 0.5 ft. and 3.0 ft. (0.15 m to 1 m) of treated wastewater, (2) the infiltration of this through the soil surface, (3) percolation through the soil matrix, and (4) eventual entrance to the ground water. Renovation of the wastewater occurs by physical, chemical, and biological processes (1).

The Federal Water Pollution Control Act Amendments of 1972, PL 92-500, has brought land treatment into the spotlight. There are four sections within which land treatment systems are to be considered (2): Section 208 - Areawide Waste Treatment Management; Section 201 - Facilities Planning; Section 204 - Best Practicable Treatment Technology; and Section 212 - Cost Effectiveness Analysis. These and other program memoranda to EPA regional administrators have resulted in the growing interest in land treatment.

The cost effectivensss analysis has shown land treatment to be a very economical treatment technique. In many cases, a true cost effectiveness analysis is impossible

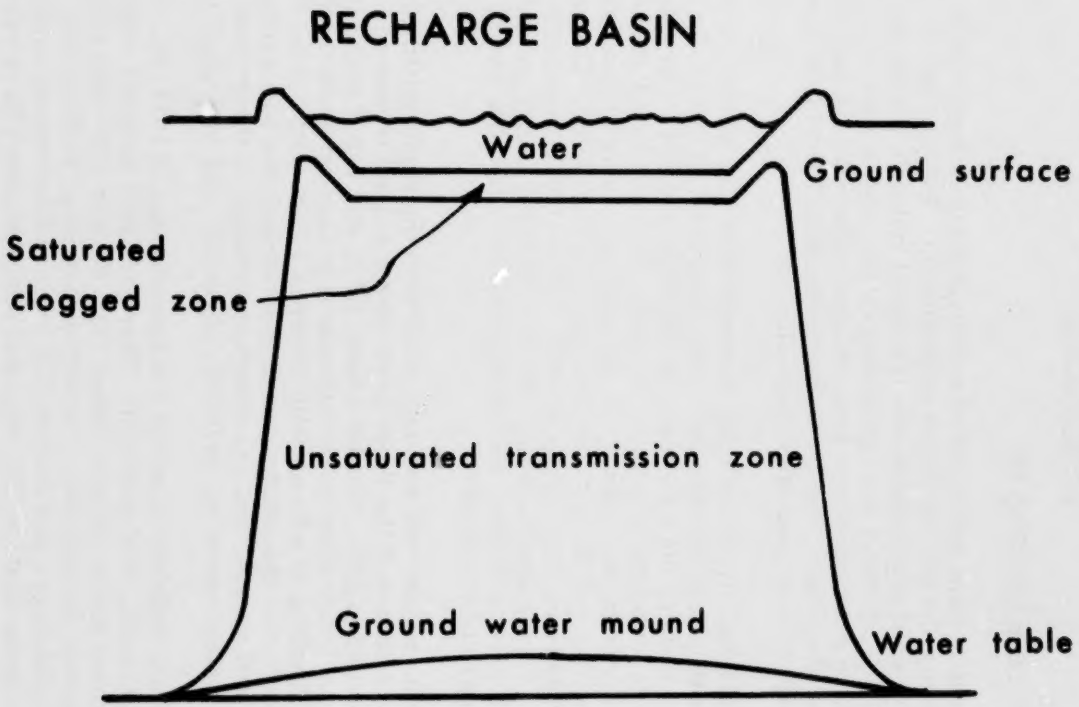

RECHARGE BASIN
Water
Ground surface
Saturated clogged zone
Unsaturated transmission zone
Ground water mound
Water table

Figure 1. Diagrammatic Sketch of the Operation of a Rapid Infiltration System for Land Application of Water and Wastewater.

to perform an 85-90 percent removal analysis. Most land treatment systems approach 100 percent BOD removal. A recent study has shown that land treatment systems are considerably less expensive, expecially in smaller treatment plants (3,4). Of the available land treatment techniques, rapid infiltration is the most economical (1). Capital and operating costs are much lower than for OF and SI due to smaller land requirements and gravity distribution. Generally, pretreatment costs are higher than for other land treatment methods.

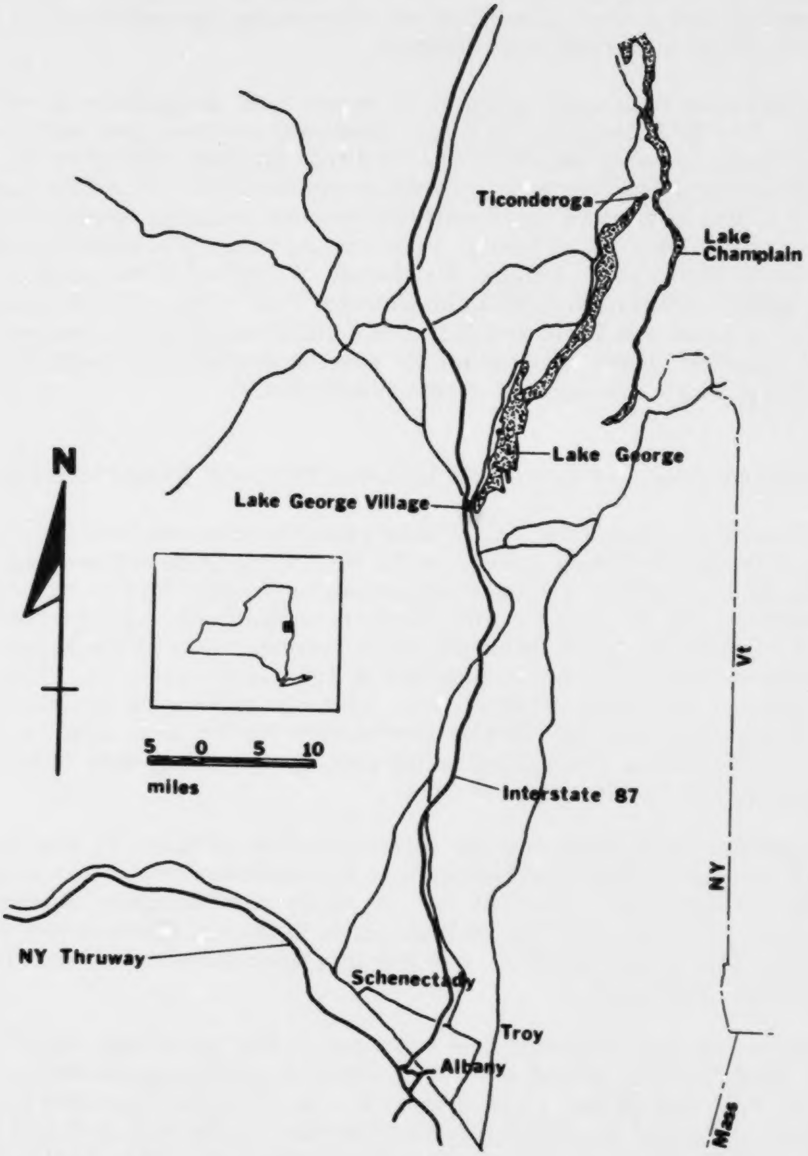
The application of sewage to the land in one form or another dates back to ancient times. The EPA has identified soil treatment systems that were established in the United States as early as 1880 (5). In 1972, the EPA identified 571 municipal wastewater facilities using surface application systems (1). The EPA has illustrated the feasibility of the RI method by identifying several existing successful facilities. These include (1): Hemet, California; Phoenix, Arizona (Flushing Meadows); Whittier Narrows, California; Santee, California; Vineland, New Jersey; Marysville, California; Westby, Wisconsin; and Lake George, New York. The effectiveness of land treatment as an available method of advanced wastewater treatment has been well documented. More work must be done on design and operational criteria in order to design such systems for optimum efficiency.

LAKE GEORGE AND THE LAKE GEORGE VILLAGE SEWAGE TREATMENT PLANT

Lake George is a beautiful, clear lake noted for its tree-lined shores. The Lake is located in the southeast corner of the New York State Adirondack Park (Figure 2). The Lake George area is a tourist attraction in summer and in winter. Most of its prosperity is due to the recreational value of the Lake. Much credit must be given to the Lake George Association for the maintenance of the Lake's purity. The Lake George Association was organized in 1885 and, due to its efforts, the Lake was given an "AA" classification (6). This classsification prohibits sewage discharges of any type into the Lake or any waters discharging into the Lake (7). The Lake water is used as a drinking water supply requiring only chlorination prior to use as public drinking water (8).

The Lake George Village sewage treatment plant (Figure 3) was constructed in 1936. The law concerning discharges into the drainage basin was interpreted to mean surface discharges. The law did not apply to subsurface discharges; therefore, soil system disposal (including septic tanks) of effluent was considered within the law. The treatment plant was put into operation in 1939 and has run continuously since that time.

The treatment plant has had five additions to the initial six sand beds since it was built. Bed N-7 was added in 1947, beds N-8 and 9 were added in 1950, and beds N-10, 11, and 12 were added in 1956. In 1965 the plant had a major expansion with eight additional beds put into service (the South beds 1-6 and North beds 13 and 14). In 1970, the final bed (S-7) was put into use. The plant is located (Figure 4) southwest of the lake. The influent scheme is an intermittent pumped system from two separate force mains located at the edge of the lake.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Lake Champlain
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Lake George Village
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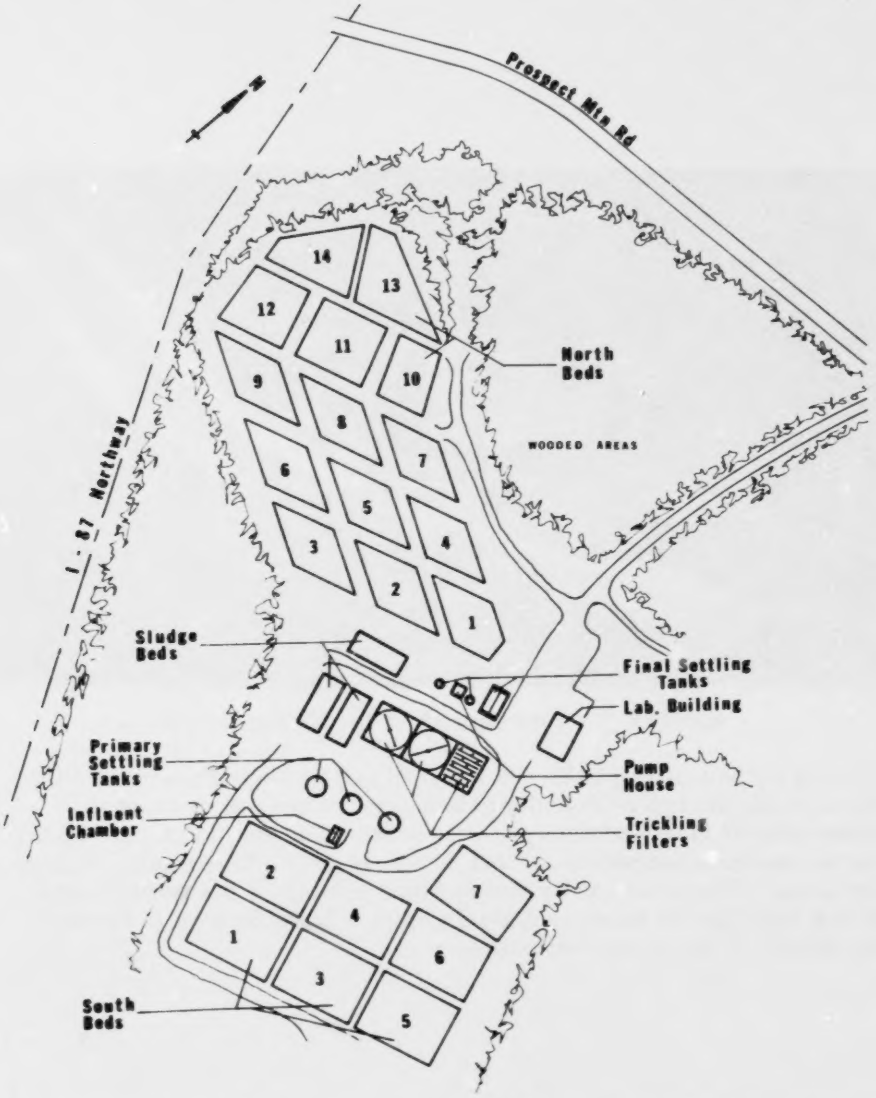
Schenectady
Troy
Albany
Mass

Figure 2. Map Showing Location of Lake George Village.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I - 87 Northway
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13
12
11
10
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1
North Beds
Sludge Beds
Final Settling Tanks
Lab. Building
Primary Settling Tanks
Pump House
Influent Chamber
Trickling Filters
2
1
4
3
7
6
5
South Beds

Figure 3. Plan of the Lake George Village Sewage Treatment Plant.

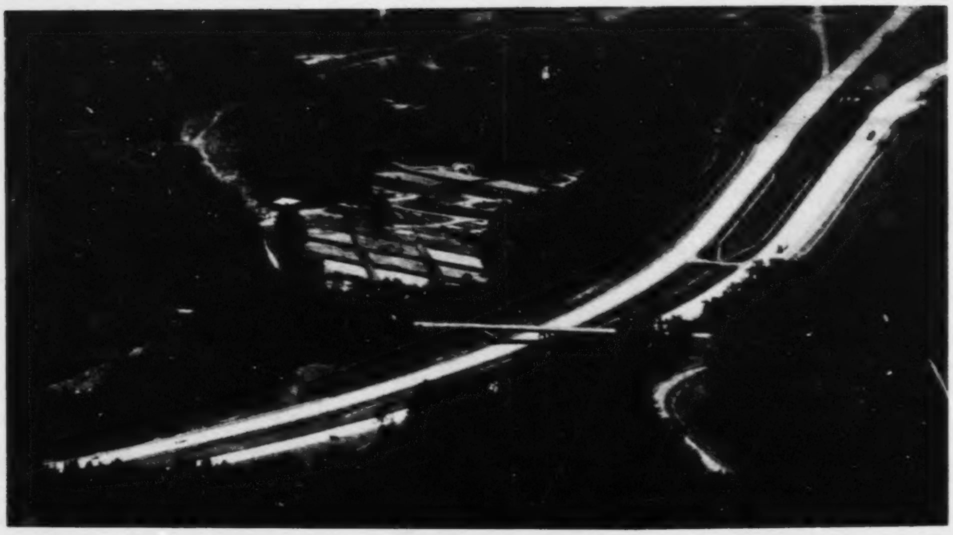


FIGURE 4. Photograph of the Treatment Plant

This is a view of the Lake George Village Sewage Treatment Plant taken from the top of Prospect Mountain. Looking southeast, Interstate 87 (the Northway), lies adjacent to the treatment plant on its western boundary. West Brook is in the foreground hidden by trees. The road in the lower right corner of the photo leads to the West Brook upstream staff gauge. Lake George is located northeast of the treatment plant.

The plant was originally built in triplicate (9). Summer flows were approximately three times the winter flows. Today the ratio is approximately two to one (10). At the head of the plant are two nine-inch (29.3 cm) Parshall flumes to record the flow of the Village and Town pumping stations. Primary treatment consists of one circular Imhoff tank and two mechanically cleaned circular settling digestion tanks (clarigesters). Secondary treatment is accomplished by two high rate rotating arm trickling filters in summer and one covered standard rate fixed nozzle sprinkling filter in winter. Secondary sedimentation is accomplished by two mechanically cleaned rectangular settling tanks and two circular settling tanks. After secondary sedimentation, the treated sewage is discharged without chlorination to the sand infiltration beds.

Gravity is used to convey the treated sewage to the 14 northern (lower) beds. The sewage is pumped to the newer 7 southern (upper) beds. Normally, the sewage is discharged to two beds at a time, one lower bed, and one upper bed. Dosing is changed at approximately 8: 00am and at 4: 00pm; thus, the entire day's flow is discharged to a total of four beds. The effluent takes from 1/2 to 3 days to seep into the ground, depending primarily on the size, age, and condition of the bed. The newer south beds are fairly uniform in size, but the areas of the north beds vary significantly. The newer beds have higher infiltration rates than the older beds. They all require periodic removal of the surface mat which forms on the sand and inhibits infiltration. There is no set cleaning schedule; the cleaning of the beds is based on the observed condition of the bed, availability of the bed for drying and cleaning, and upon the time spent by plant personnel on other duties. Cleaning consists of raking and removing the upper few cm of sand and discing followed by releveling of the sand surface. The mat which is removed is either taken to a sanitary landfill or it is applied on the forest floor in areas surrounding the treatment plant. Sludges from the settling tanks are dried on sludge drying beds and are then disposed of in a sanitary landfill.

Since this system has been in operation for 38 years, it affords an ideal opportunity to study the long term effects of treatment using the rapid infiltration technique. Many other areas may find such systems attractive. It is hoped that this study of the Lake George system may help resolve problems in the design of future rapid infiltration systems.

SECTION 2

CONCLUSIONS

The application of secondary treated effluent from a conventional domestic sewage treatment plant onto natural delta sand beds using the rapid infiltration technique with intermittent dosing has been shown to provide the equivalent of tertiary treatment. Such a system may be considered to be continuously effective for long periods of time as exemplified by nearly 40 years of operation of the Lake George Village Sewage Treatment Plant.

A significant portion of the purification of the secondary effluent applied to the sand beds was accomplished in the upper unsaturated zone of the sand infiltration beds during the vertical transport of the liquid. Biological oxygen demand (BOD), chemical oxygen demand (COD), coliforms, fecal coliforms, and streptococci were effectively removed in the top 3 m (10 ft) of sand. Ammonia and organic nitrogen were oxidized to nitrate in the top 3 m (10 ft) of sand. Data indicated a further reduction of nitrate at greater depths, with almost total removal of nitrate by the time the wastewater passed through 18 m (60 ft) of sand. There appeared to be a positive correlation between DO, redox potential, and the oxidation of reduced nitrogen compounds to nitrate, followed by the reduction of nitrate to nitrogen gas. This correlation is supported by measurable DO present at all depths within the sand beds and within the aquifer below the sand beds to depths of 18 m (60 ft) where reduced nitrogen levels were observed. There was further oxidation of reduced nitrogen compounds in the horizontal transport of these compounds through the soil. Orthophosphate was generally reduced to concentration less than 1 mg/l as P in 3 m (10 ft) of vertical transport, and concentrations seldom exceeded 0.2 mg/l at the 7 m (23 ft) depth. There was further removal of phosphate in the horizontal transport of phosphorus compounds in the saturated aquifer below the sand beds. Where the ground water reappears as seepage, approximately 600 m (2,000 ft) from the sand infiltration beds, the soluble phosphate content averaged 14 μg/l which is approximately the same as the natural ground water in the area.

Non-reactive soluble substances such as sodium, chloride, potassium, and total dissolved solids showed no significant changes during vertical or horizontal transport through the sand.

Low concentrations of viruses (coliphage) were found in all the sampling wells including the control wells. Highest values were found in wells 1 and 9, but there was no particular pattern with respect to well location and concentration in any of the other wells. Insufficient information was available to evaluate the significance of these viruses or the degree of removal of them.

Copper and aluminum were found to be present in concentrations lower than the detectable limits with normal atomic adsorption techniques. The iron concentration in the ground water receiving sewage effluent was lower than in the natural ground water in the surrounding area. Anionic non-soap surfactants were below detectable limits at all locations.

Although there was little significant change in pH with either vertical or horizontal movement through the soil, there was an increase in alkalinity with depth in the sand beds. This may be accredited to changes in the carbonate-bicarbonate balance caused by production of CO_2 by biological activity within the sand beds.

Intermittent dosing of the sand beds is important in maintaining both efficiency of treatment and speed of infiltration of the liquid onto the soil. Actual infiltration rates ranged from a low of 0.07 m/day (0.25 ft/day with a head of 0.15 m (0.05 ft) of sewage on the bed to a high of 0.7 m/day (2.25 ft/day) with a head of 0.46 m (1.5 ft) of effluent on a clean bed. Infiltration rates with a bed covered with weeds were lower than on a weed-free bed when the depth of liquid on the sand bed was less than 0.3 m (1 ft) and higher on a weed-free bed when the loading depths exceeded this amount. The average vertical velocity in bed N-11 was .63 m/day (2 ft/day). The horizontal velocity varied throughout the area but was in the general range of 9 m/day (30 ft/day).

SECTION 3

RECOMMENDATIONS

For small installations treating primarily domestic sewage, a rapid infiltration system applying secondary treated effluent to sand with intermittent dosing is recommended where site conditions are favorable. This system could provide an efficient, inexpensive, and energy conserving method of tertiary treatment for the removal of most common contaminants and nutrients. Although each individual situation should be monitored routinely, such a system can be expected to last for many years and perhaps indefinitely.

It has been shown that phosphorus may be completely removed in a sand infiltration system, even though mechanisms of removal are not understood. Additional studies should be made to determine the methods of removal so that this system could be applied in other similar situations.

Complete nitrogen removal appears to be accomplished in the deeper (18 m) sand beds. Additional studies should be made to determine if more complete removal of nitrogen could be achieved in less vertical distance. Methods to vary the DO and redox potential may be effective in achieving nitrogen removal. In addition, the removal of nitrogen by plant extraction, such as allowing the weeds to grow on the sand beds, should be studied.

SECTION 4

MATERIALS AND METHODS

The basic facilities used for this study were the existing Lake George Village Sewage Treatment Plant (Fig. 3) and its variable inflows from Lake George Village and the Town of Lake George. The sewage influent to the treatment plant is primarily of domestic nature received from both the Village and Town of Lake George which are highly seasonal tourist areas. The flows have increased over a period of years as shown in Fig. 5 with the summer flows presently about double the winter flows. The peak daily summer flows reach approximately 5×10^6 l/day (1.3 mgd). The plant was put into operation in 1939 (9) to meet the requirements of the state regulations (7) that there should be no discharge of sewage or sewage effluent into Lake George or any tributary of Lake George. It was considered that the application of the treated effluent into sand beds would provide sufficient treatment that by the time the water did reach Lake George it would have no effect upon the quality of the Lake.

The treatment plant has operated continuously since its completion; however, no studies had been conducted to determine the quality of the effluent or the final destination of the applied effluent in the sand beds until an initial attempt was made by the author in 1968 (11). These initial studies were begun even though sufficient resources were not available for a detailed study at that time.

In order to determine the possible direction of flow of the treatment plant effluent infiltrated into the ground through sand beds, additional studies were conducted during 1972 (12) in which the resistivity of the soil including the ground water was measured. The general direction of low resistivity (high conductivity) was observed to follow a path nearly due north from the treatment plant along Gage Road toward West Brook. The study could not conclude whether or not the zone of low resistivity passed under West Brook or terminated at this location.

Based upon the above information, a reconnaissance survey was conducted of the area along the south bank of West Brook. Considerable seepage was observed coming out of the base of the hill at the edge of the flood plain of West Brook. The conductivity of the seepage was high, giving the first indication of the possibility that the applied sewage effluent was reappearing from the ground and becoming surface water which flowed into West Brook and ultimately into Lake George. With this information available, the New York State Department of Environmental Conservation (NYSDEC) began installing observation and sampling wells in several of the sand beds and in the area between the sand beds and West Brook as the need indicated to sample the water quality treatment efficiency of the sand beds. The locations of all the wells and other sampling points in the area are shown in Figure 6. Wells 4, 7, and 10 were designed to be control wells unaffected by the applied

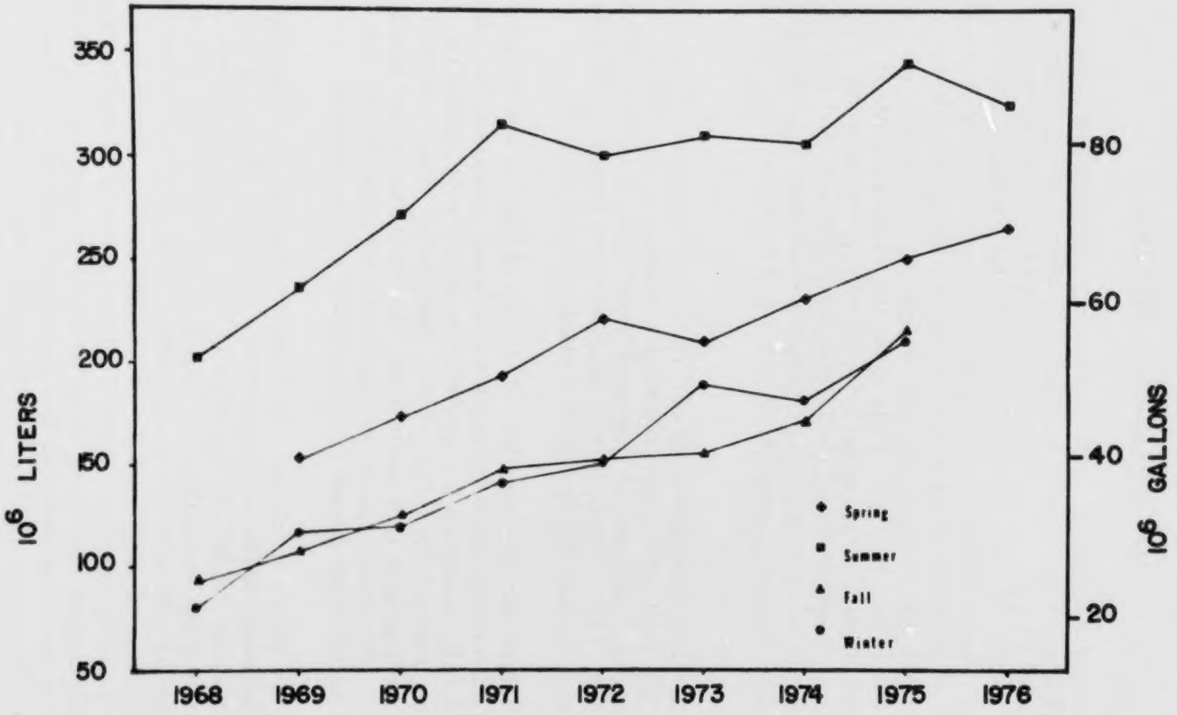


Figure 5. Seasonal Flows Through the Lake George Village Sewage Treatment Plant.

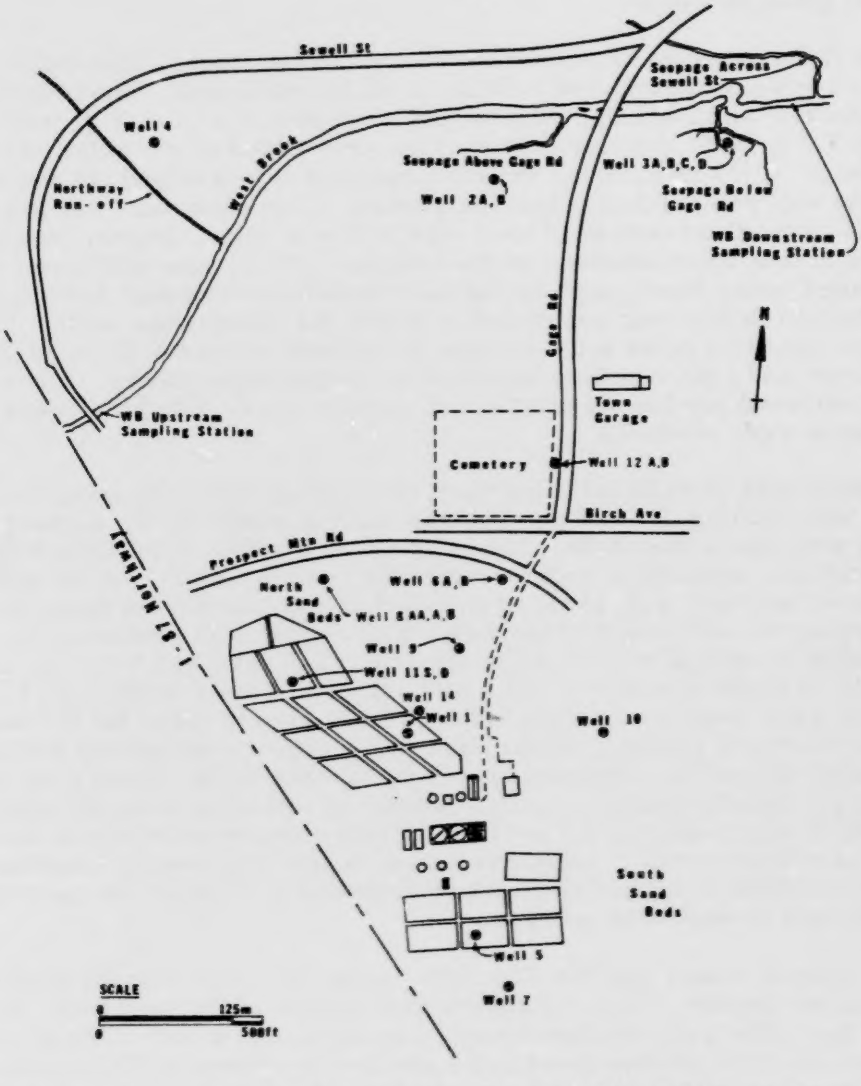

Sewell St
Seepage Across Sewell St
Well 4
West Brook
Seepage Above Gage Rd
Well 3A,B,C,D
Northway Run-off
Well 2A,B
Seepage Below Gage Rd
WB Downstream Sampling Station
Gage Rd
N
Town Garage
WB Upstream Sampling Station
Cemetery
Well 12 A,B
Birch Ave
I-87 Northway
Prospect Mtn Rd
North Sand Beds
Well 6A,B
Well 8 AA,A,B
Well 9
Well 11 S,D
Well 14
Well 1
Well 10
South Sand Beds
Well 5
Well 7
SCALE
0
125m
0
500ft

Figure 6. Map of the General Area Involved in this Study Showing the Observation Wells and Other Sampling Points.

sewage effluent, and therefore indicative of the quality of the natural ground water in the area. An attempt was made to locate one additional control well, no. 13, along Prospect Mt. Road on the eastern side of Rt. 187 (the Northway); however, an obstruction was encountered before the ground water table was penetrated at this location. The elevations of the ground surface, ground water level, well screen elevations, and the bedrock elevations of each well are compared in Fig. 7 with actual data given in Table 1.

Three different types of wells were utilized in this study. The wells marked S in Table 1 were constructed of 3.15 cm (1.25 in) steel pipe. The first section of pipe consisted of a steel tipped point and a 0.6 or 0.9 m (2 or 3 ft) well screen. Additional 1.3 m (4 ft) sections of drive pipe were added as the point was driven into the sand. A 113 kg (250 lb) weight suspended from a tripod rig was used to drive the well point and pipe into the ground. This equipment was provided by NYSDEC. One disadvantage of steel pipe is that it rusts, thereby invalidating any values of iron determinations in the samples. Thus, some additional wells were installed using plastic pipe to eliminate sample interference from rust. A commercial well driller was contracted to install the plastic pipe wells. Ten centimeter (4 in) diameter holes were augered to bedrock and the 3.15 cm (1.25 in) plastic screen and pipe was then installed inside the auger tubing. The auger was then retrieved leaving the plastic well pipe in place. The holes were backfilled after the tools were retrieved.

The third type of wells installed were constructed using the cable-tool precussion method of well drilling (13). This involved raising and dropping a heavy string of drilling tools into a bore hole. The reciprocating action of the tools mixed a slurry which was removed at regular intervals using a bailer. As the soil was loosened and removed, a 15.25 cm (6 in) steel casing was driven down the bore hole to support the side-walls of the well. Within this casing submersible pumps were installed in each of the two wells constructed in sand bed N-11 and are labeled 11S and 11D (shallow and deep). The well screens in these wells were 0.9 m (3 ft) long and were located in the top 0.9 m (3 ft) of ground water for the shallow well and at the top of bedrock for the deep well. Appropriate wiring and piping were installed to provide continuous pumping of these wells. During the installation of these large diameter pumping wells, samples of soil were taken for analysis at intervals of approximately 0.6 m (2 ft). These samples were sent to the U. S. Army Corps of Engineers Cold Regions Research and Engineering Laboratory, Hanover, New Hampshire, for chemical analysis and to NYSDEC for particle distribution and permeability analysis.

In the special case of bed N-11, suction lysimeters were also installed for sampling at various depths within the unsaturated portion of the sand bed. Holes were drilled by the cable-tool percussion method as described above. The suction lysimeters were lowered into the cased holes attached to a piece of PVC pipe for support. The holes were backfilled with a slurry of 100 mesh silica sand placed around the porous ceramic cups of the lysimeters followed by a layer of sand that had been removed from the bore hole, as shown in Fig. 8. A plug of bentonite clay was placed between each successive lysimeter to prevent the direct passage of any liquid down through possible voids surrounding the lysimeter. The casing was retrieved as backfilling progressed. Several of the lysimeters sank when a section of casing was removed, resulting in their exact depth not being accurately known. Of the 13 lysimeters installed, only four were found to operate satisfactorily.

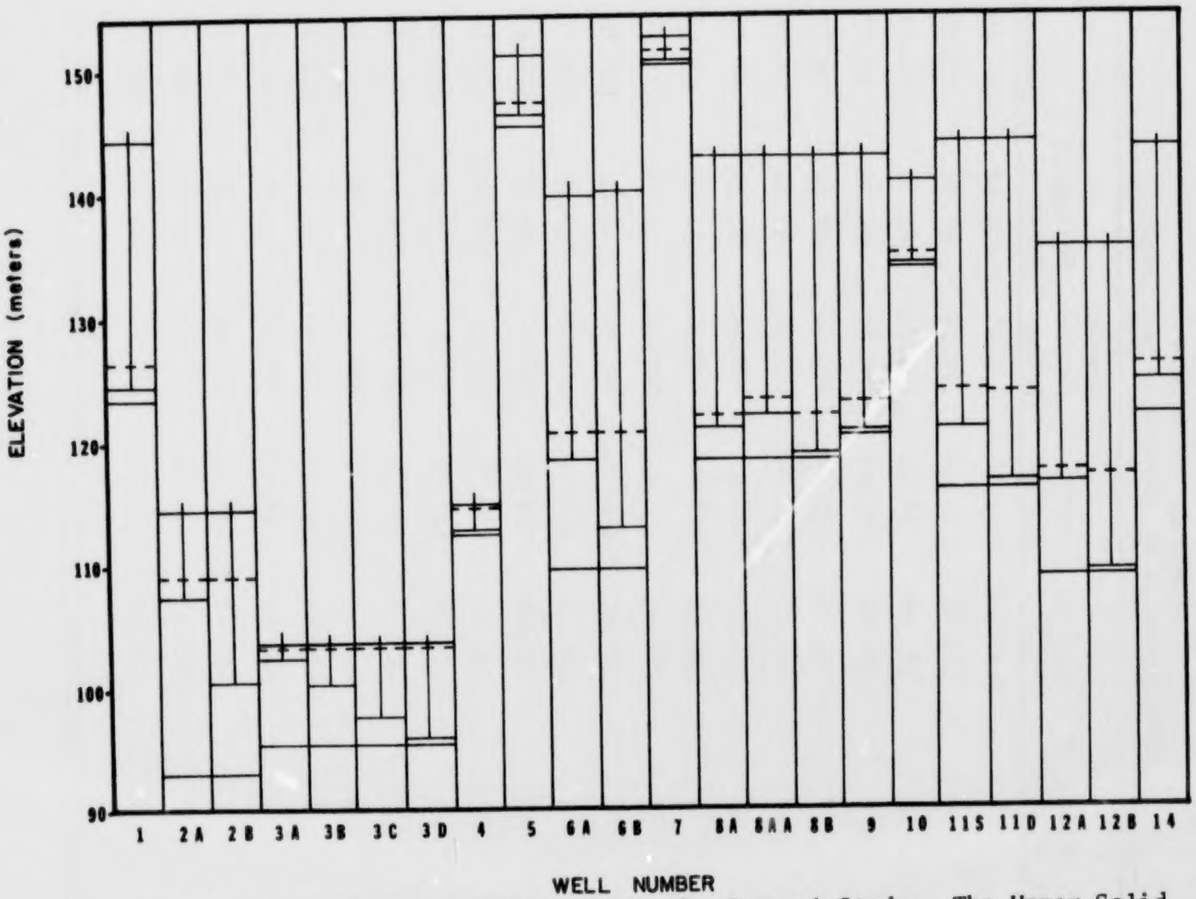


Figure 7. Sketch of the Wells Used in the General Study. The Upper Solid Line Indicates the Ground Surface Level; the Dashed Line Indicates the Approximate Water Level in the Well; the Next Solid Line at the Bottom of the Vertical Line Indicates the Depth of the Well Point, the Lowest Solid Line Indicates the Depth of the Bedrock.

TABLE 1. WELL DATA
(Elevations in m. Above Mean Sea Level)

Location	Steel or Plastic	Top of Well	Ground Surface	Approx. Ground Water	Bottom of Point	Bedrock
1	P	145.57	144.77	126.47	124.44	123.48
2A	P	115.25	114.45	109.28	107.36	93.29
2B	S	115.44	114.45	109.31	100.72	93.73
3A	P	104.05	103.63	103.47	102.57	95.73
3B	P	104.19	103.63	103.52	100.32	95.73
3C	P	104.35	103.63	103.54	97.94	95.73
3D	S	103.73	103.63	103.44	96.23	95.73
4	S	115.68	114.52	114.50	112.65	112.65
5	P	152.50	151.03	147.22	146.16	145.55
6A	P	140.03	139.73	120.92	118.68	109.76
6B	S	140.25	139.85	120.59	109.76	109.76
7	S	153.85	152.53	151.55'	150.47	150.47
8AA	P	143.37	143.30	123.48	121.96	118.38
8A	P	143.66	143.21	122.03	121.21	118.38
8B	P	143.68	143.29	122.02	118.38	118.38
9	S	143.11	142.28	123.10	120.58	120.58
10	S	141.80	141.08	134.66	133.81	133.81
11S	S	144.98	143.95	124.06	121.09	115.91
11D	S	145.17	143.95	123.81	117.40	115.91
12A	P	136.04	135.69	117.54	116.55	109.05
12B	P	136.06	135.71	117.04	109.07	109.07
14	S	144.27	143.80	125.81	124.61	124.00

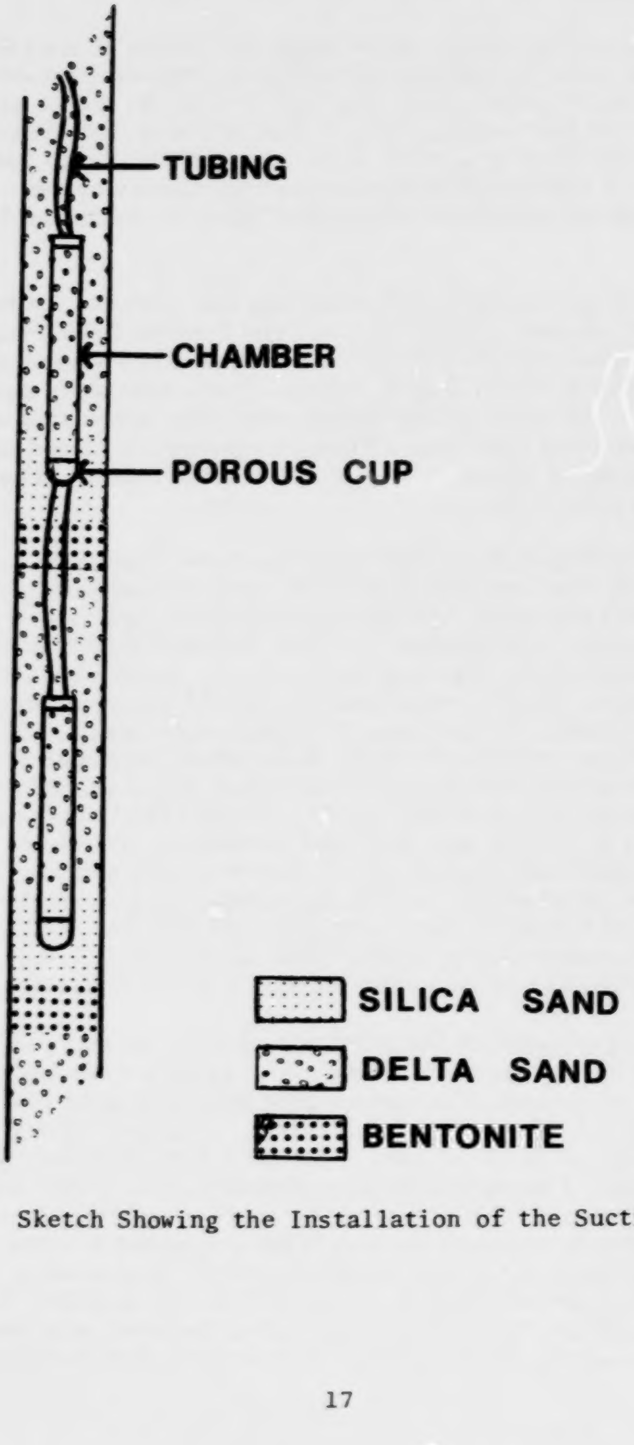

TUBING
CHAMBER
POROUS CUP
SILICA SAND
DELTA SAND
BENTONITE

Figure 8. Sketch Showing the Installation of the Suction Lysimeters.

The actual location of the wells and lysimeters in Bed N-11 is shown in Fig. 9, and the depths of the individual wells and lysimeters are shown in Fig. 10. Chemical analysis presented later indicates that the 4 operating lysimeters were located at depth intervals sufficient to obtain representative information on the change of quality with depth within the sand bed.

The two seepage areas which occur along the bottom of the hill on the south bank of West Brook were probably originally one large seepage area that has been split by the construction of the road bed of Gage Road. A small stream represents the total seepage which occurs above Gage Road; below Gage Road the seepage occurs over a wider area. Flow of this large seepage area was channeled into a common ditch by connecting numerous areas of seepage into one main stream so that the seepage flow could be monitored before entering West Brook.

In order to measure the flow of the two seepage streams, a small dam was constructed in each stream, and a Stevens Type F water level recorder was installed. Cross-sectioned flow measurements were made a number of times for each stream using a Gurley Pigmy meter. These data were used to construct stage rating curves for each stream which were then correlated with the stage of the Stevens water level recorder. Thus, a continuous record of flow was provided with flow-depth charts from the Stevens recorders. These charts were changed on a weekly basis.

Tracer studies using both tritium and rhodamine dyes were conducted to follow both the vertical and horizontal movement of sewage plant effluent discharged onto the sand beds. An initial study using rhodamine B dye was conducted without success. Permission was then obtained from the NYSDEC Bureau of Radiation to conduct radio tracer studies. In the tracer studies, both tritium (radioactive hydrogen-3) and rhodamine WT (as 20% solution) were used as tracers. Three 0.1 curie (Ci) portions of tritium were used, one for each of three separate tracer studies. In order to differentiate between the three tritium tracer studies, a tertiary tracer was added to the second and third studies. In the second tritium tracer study, 108 kg (240 lb) of sodium chloride were dissolved in a 379 1 (100 gal) tank and introduced into North Sand Bed 4 along with the tritium and rhodamine WT tracers. The amount of sodium chloride added was calculated to double the normal chloride content of the sewage. In the third tritium tracer study, 22.7 kg (50 lbs) of KCl were added as described in the second tracer study. This amount of KCl was designed to double the concentration of potassium normally found in the applied liquid.

Special sampling equipment designed specifically for this project included a bailer, a vacuum system, and a scooper. The bailers were custom-made to fit the wells. They consisted of copper pipe approximately 1.52 to 1.83 m (5-6 ft) long with a 1.91 cm (3/4 in) diameter. A plastic check valve was installed at the bottom of the pipe, and holes were drilled at the top in order to attach a nylon cord. The cord was approximately 24 m (80 ft) long ,thus allowing the bailer to reach the bottom of the deepest wells. The vacuum system consisted of a 1000 ml Erlenmeyer vacuum flask connected to either a hand vacuum pump or an upright vacuum-pressure pump. The scooper was made of an aluminum pole approximately 1.91 cm (3/4 in) in diameter, 1.52 to 1.83 m (5-6 ft) long, with an aluminum cup at the bottom fastened with two screws. This scooper was used to collect samples of treatment plant influent.

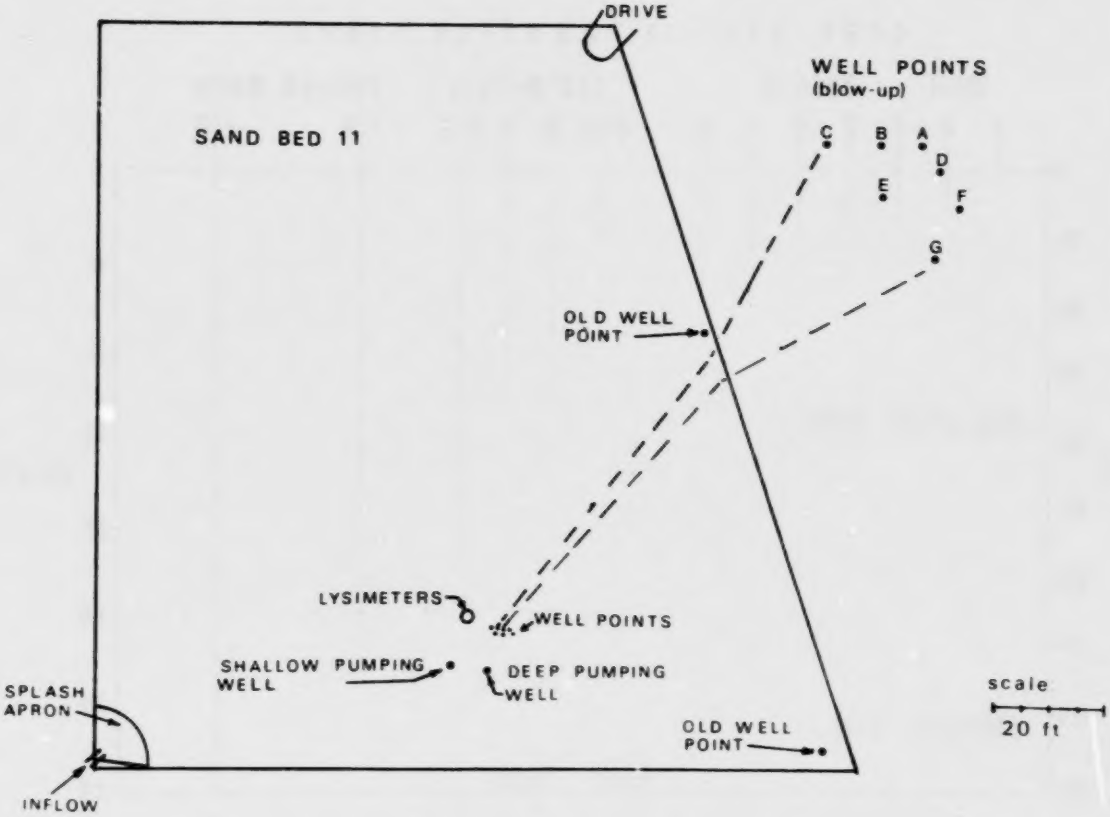

DRIVE
WELL POINTS
(blow-up)
SAND BED 11
C
B
A
D
E
F
G
OLD WELL POINT
LYSIMETERS
WELL POINTS
SHALLOW PUMPING WELL
DEEP PUMPING WELL
SPLASH APRON
scale
20 ft
OLD WELL POINT
INFLOW

Figure 9. Plan of Bed N-11 Showing the Location of the Driven Well Points, the Lysimeters and the Shallow and Deep Pumping Wells.

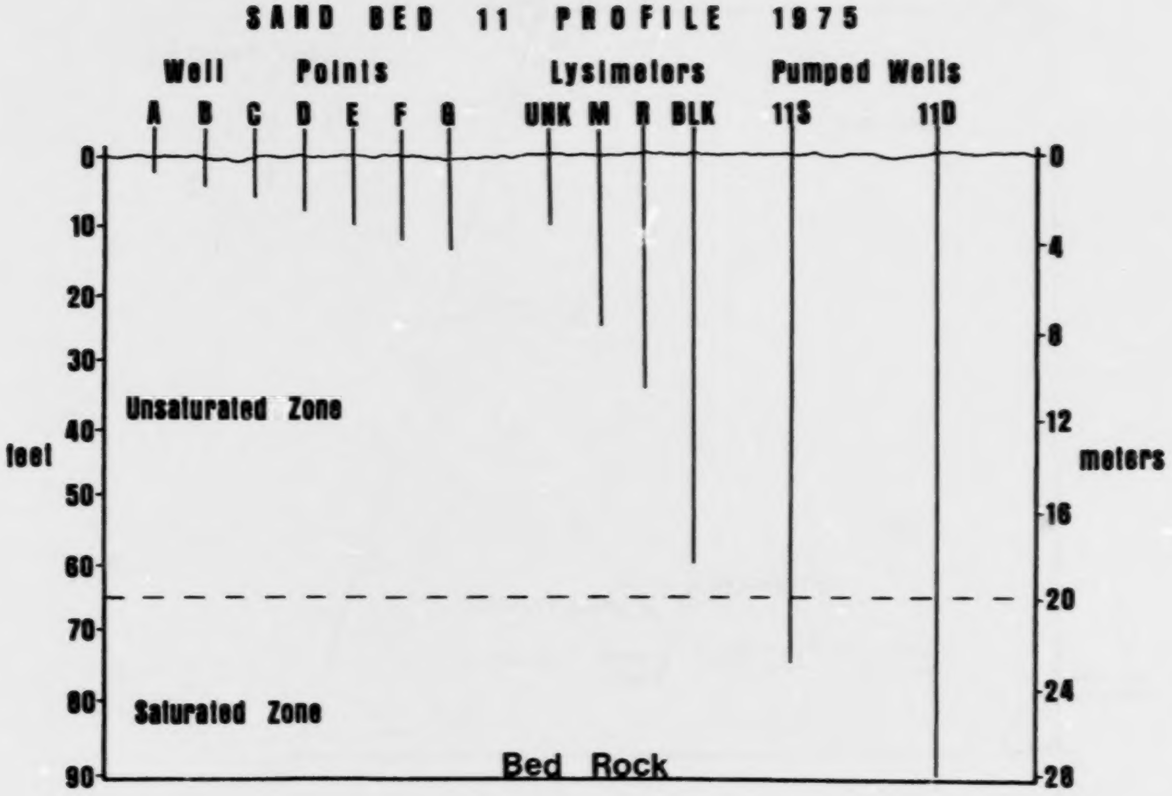


Figure 10. Profile of Bed N-11 Showing the Depth of the Driven Well Points, the Operational Lysimeters and the Shallow and Deep Pumped Wells.

The water level in each well was measured using an electrical water-sensing depth gauge. This device was also used to find the elevation of the bottom of the well points by lowering the sensing probe into the well until it reached bottom. Mean sea level elevations for tops of well pipes, ground surfaces, water levels in wells, and bottoms of wells were established.

SECTION 5

EXPERIMENTAL PROCEDURES

SAMPLING PROCEDURES

Intense efforts were made to secure representative water quality samples of both ground waters and surface waters of the Lake George Treatment System. Samples were collected from the inflow of raw sewage until it emerged from underground and entered into West Brook. Monitoring was conducted through all seasons including two harsh New England winters when snows at the site exceeded depths of 0.6 m (2 ft), and temperatures were well below -10°C for several weeks. Monitoring included 13 well locations, some of which were well clusters from which samples were collected from as many as four depths. Other sampling points included two seepage areas (one upstream and one downstream of Gage Road), West Brook (both upstream and downstream of the two seepage areas), the raw influent to the plant, and effluent discharged to various sand beds. The complete monitoring network included 28 separate water quality monitoring locations.

Ground water samples were obtained from both the observation wells and the pumping wells. In the observation wells, samples were collected using either a bailer or a vacuum pump depending upon the depth to water in the well. The volume of sample collected with the bailer at each dipping was approximately 500 ml. The contents of the bailer were emptied into labelled plastic containers which were first rinsed with the sample. The dipping was repeated as often as necessary to secure a sufficient sample volume.

The vacuum pump was used for the shallower wells, especially wells 2A and B and 3A, B, C, and D. A plastic sampling tube was lowered into the well, and a vacuum was applied to the attached suction flask. After approximately 200 ml of sample was collected, the flask was rinsed, and a sample was collected for analysis.

Surface samples were collected at West Brook, the seepages, and the sewage influent and effluent. At West Brook and the seepages, the plastic sample containers were first rinsed with the sample and then simply filled by dipping the container into the sample source. The sewage influent was collected using the scooper described earlier.

A pressure-vacuum pump was utilized for securing samples from the suction lysimeters. The lysimeters were designed with two tubes (Fig. 8); one for retrieving the sample and the other to apply a pressure to force the sample from the porous cup or to apply a vacuum to enable the porous cup to become filled. To maintain a vacuum, a C-clamp was attached to the exposed end of each tube. To secure a sample the C-clamps were removed from both tubes, and pressure was applied to the pressure-vacuum tubing, forcing the sample through the collection tube. After removing the sample, the lysimeter was prepared for sample collection by placing the C-clamp on the delivery tube and drawing a vacuum on the pressure-vacuum tube. A C-clamp was then placed on the vacuum tube before the vacuum pump was detached leaving a negative pressure in the lysimeter. Varying amounts of sample were secured from the lysimeters depending on the time between sampling, the ground conditions, and the water content of the soil system, which in turn was a function of bed dosing. Volumes of samples varied from a few ml to as much as 600 ml. Due to the scarcity of samples, the sampling containers were not rinsed prior to filling. Special precautions were taken to prevent contamination of sample containers for the lysimeters.

Wells 11S and 11D were generally sampled by means of submersible pumps. The pump was operated for 5 to 10 minutes prior to sample collection to assure that a fresh sample was being collected from the aquifer. During the period from July 25, 1975, until May 25, 1976, while the submersible pump on well 11S was inoperable, samples were collected using a bailer.

Plastic containers (1000 ml) were used to collect samples. Three containers were filled for each sample. Two containers of sample were preserved with 40 mg $HgCl_2$/l for nutrient analysis as described in Standard Methods (14). Nutrient analysis was conducted at the Fresh Water Institute at Lake George. Analyses were performed for orthophosphate, total phosphorus, nitrate nitrogen, ammonia nitrogen, and total Kjeldahl nitrogen. The third sample was stored at 4°C and taken to the Rensselaer Polytechnic Institute, North Hall Laboratory, for the determination of alkalinity, chloride, biochemical oxygen demand (BOD), calcium, metals and other constitutents. All samples were prepared and run as quickly as possible after arrival at the North Hall Laboratory. Separate sterile containers were used to collect bacteriological samples. These samples were collected simultaneously with all other samples.

Special difficulties were encountered when sampling during subfreezing temperatures. A temperature of -33°C (-27°F) was recorded during one sampling trip. When temperatures became this low, problems arose with the sampling equipment and the sample. Frequently the water samples froze as soon as they came in contact with the ambient air. Ice clogged the sampling tubes on the vacuum sampler several times. Another problem encountered was freezing of the observation wells. It was learned later in the project that these wells had less tendency to freeze if the cap was left off the top allowing the water vapor generated from the aquifer to escape, thereby keeping the well ice free above the frost line. Heavy snowfall created a problem of access to some of the wells such as 2, 6, and 10, resulting in sparse data for these wells during winter monitoring.

ANALYTICAL PROCEDURES

Field Analyses

Tests for unstable parameters, such as temperature and dissolved oxygen (DO), were performed in the field. Other field measurements include dissolved solids, redox potential, and pH. Temperature and DO were measured using a Yellow Springs Model 54 Oxygen Meter with a probe on a 30 m (100 ft) cable. This portable instrument allowed in situ measurements in all wells and surface samples. The probe was maintained and operated according to the manufacturer's instructions.

A TRI-R dissolved solids meter reading in parts per million was used to determine dissolved solids concentration. The sample cell was rinsed with sample prior to measurement.

A Leeds and Northrup No. 7417 portable pH meter was used to determine pH and oxidation-reduction (redox) potential in the field. A platinum electrode No. 117225 was required for redox measurements. Due to technical problems, this meter was only used near the end of the sampling schedule.

Ground water elevations of the sample wells were determined using a portable electric depth gauge (Soil Test, Inc. Model DR-760A). This instrument consists of an ammeter, which records the closed circuit of two electrodes on a sounding cable lowered down the well bore when contact is made with the water surface. The cable is calibrated in 1.5 m (5 ft) increments, and a steel tape was used to obtain accurate measurements between the increments and the top of the well pipe.

The frequency of sampling was a function of the sample locations, the specific study being conducted at the time, and the season. In general, samples were collected from the observation wells biweekly, with half of the locations being sampled on alternate weeks. Samples from the lysimeters were usually collected weekly, but during some periods, samples were collected three times per week so that enough sample volume could be accumulated for analysis. Tracer study samples were collected daily in observation wells surrounding sand beds where tracers were introduced. In the tracer studies conducted in bed N11, the submersible pumps were operated continuously supplying samples to a Turner Model 111 Fluorometer equipped with a flow-thru sampling door and a continuous recorder. Stream flow measurements were also monitored continuously with Stevens Type F water level recorders. Flows of the treatment plant influent were monitored by means of the continuous flow recorders which are part of the treatment plant.

Laboratory Analyses

The potentiometric method described in Standard Methods (14) was used to obtain accurate readings of alkalinity. A Fisher Accumet Model 320 Expanded Scale Research pH Meter was used to determine the end points.

The Technicon Auto Analyzer II was used to determine all forms of nitrogen and phosphorus due to the large number of samples analyzed. The component parts of the system included an Auto Analyzer II, sampler, pump, manifold, S. C. Colorimeter, and Recorder. Technicon Methods Sheet No. 155-71W (Nov. 1971) was followed for determinations of soluble reactive phosphorus after filtering through a 0.45 μm membrane filter. In analysis for total soluble phosphorus, the samples were first filtered and the persulfate digestion procedure described in Standard Methods (14) was used to convert the total phosphorus to soluble reactive phosphorus and then analyzed as above.

Nitrate nitrogen and ammonia nitrogen concentrations were determined as outlined in Technicon Industrial Methods Sheet Nos. 158/71W/preliminary (Dec. 1972) and 154-71W/tentative (Feb. 1973), respectively. Kjeldahl nitrogen samples were first digested to convert the various organic forms to ammonia as in Standard Methods (14), and then the procedure for ammonia nitrogen was followed.

The membrane filter procedure outlined in Standard Methods (14) was used for determining fecal coliform concentrations. All sample bottles were sterilized in a Dispatch Oven, Style 287, Type 3-H hot air sterilizing oven at 170°C for a minimum of two hours. Graduated cylinders and medium containers were sterlized in a Wilmot Castle Co. #19465 autoclave for 15 minutes at 120°C. Presterilized disposable 60x15 mm plastic petri dishes were used. All pipets used were also the presterilized disposable type. A Gelman Manifold #4201 was used to hold three Gelman Membrane Filter Funnels. These funnels were also autoclaved at 120°C for 15 minutes for sterilization. Gelman presterilized 47 mm diameter 0.45 μm gridded filters with pads were used. The medium was BBL-M-F broth. All analyses were carried out shortly after arrival at the North Hall Laboratory. After sample preparation and filtration, the petri dishes were placed in weighted plastic bags. These plastic bags were then immersed in a Blue M Magni-Whirl Constant Temperature Bath at 44.5 ± 0.2°C for 24 hours. The colonies were counted using a Quebec Colony Counter (Spencer Lens Company).

Total coliform analyses were conducted according to procedures outlined in Standard Methods (14) which was basically the same as that for fecal coliforms. The single step direct technic was used. The medium was BBL-M-endo broth. The plates were incubated at 35 ± 0.5°C in Eimer & Amend Type II incubator for 24 hours. High humidity was maintained by placing an open beaker of distilled water in the oven. The colonies were counted with a 10x binocular microscope.

Difco M enterococcus agar was used as a selective medium for recovering fecal streptococci species. Two milliliters of sample were carefully transferred using sterile disposable pipets into 15x100 mm glass petri dishes which had been sterilized by autoclaving for 15 minutes at 120°C. Next, 5 ml of agar were transferred into the dish, again using a sterile pipet. The sample and agar were well mixed and allowed to harden. The plates were then inverted and incubated in an Eimer & Amend Type II incubator at 35 ± 0.5°C for two days. The streptococci species appeared as pink-to-red circular colonies, and they were counted using a Quebec Colony Counter (Spencer Lens Company).

The double layer agar technique of Hershey et al (15) was used for the determination of viral concentrations. Five percent chloroform was added to 50 ml of sample, and the mixture was stirred thoroughly for 30 minutes. Air was then bubbled through the mixture to remove the chloroform. The samples were then assayed according to Hershey et al (15). Pure culture of E. coli B was supplied by the Biology Department of RPI. The top agar consisted of 10 g Difco Bacto-Tryptone, 6 g Difco Bacto-Agar, 8 g NaCl, 1 g glucose, 1.5 ml of 1 N NaOH, and 2 ml of 1 N $CaCl_2$ per liter. The bottom agar contained the same ingredients, but the Bacto-Agar was increased to 10 g/l. All media and glassware were sterilized at 120°C for 15 minutes. After preparation, the glass plates were inverted and incubated at 35 ± 0.5°C for approximately 8-10 hours. Plaques were then counted with a Quebec Colony Counter (Spencer Lens Company).

Five day BOD's were run as detailed in Standards Methods (14). The only samples for which dilution was necessary were the treatment plant influent and effluent. The Azide Modification of the Winkler Method (14) was used for determining dissolved oxygen concentrations. The samples were incubated at 20 ± 1.0°C in a Hythermco thermostatically controlled incubator. In addition to the normal sampling sites, a series of samples was taken of the supernatant liquid remaining on a sand bed before it was infiltrated into the ground. These analyses were done to determine the BOD_5 reduction due to natural aeration while the effluent remained on the bed prior to infiltration.

The chloride ion concentration was determined by the Argentometric method as described in Standard Methods (14) and by the specific ion electrode method. The latter was quicker and gave results consistent with the first method. A Beckman Century S S pH meter was used with an Orion Analyzer double junction electrode, Model 90-02. The inner chamber was filled with standard Orion 90-00-02 filling solution. The outer chamber contained 4 M NH_4NO_3. Orion Research specific ion electrode model 94-17A was used as the chloride electrode. Before each set of analysis was run, the meter was calibrated to a standard curve.

Aluminum concentrations were determined with the use of a Perkin-Elmer 403 Atomic Absorption Spectrophotometer equipped with a nitrous oxide burner while iron concentrations were determined according to the tripyridine method, outlined in Standard Methods (14). The tripyridine method is subject to interferences, as is the phenanthroline method; however, there is no interference from phosphates and heavy metals in concentrations which might be expected in the samples tested.

The anionic surfactants (linear alkylate sulfonate, LAS) concentration was determined according to a recently developed, indirect two-phase titration method (16). The method uses cetyldimethylbenzyl ammonium chloride (CDBAC) in excess to form a complex with the water-soluble anionic surfactant, and uses methyl orange (MO) as an indicator in the presence of chloroform. The color of the CDBAC-MO complex in the chloroform phase is yellow. This water-chloroform two-phase mixture is then titrated with sodium tetraphenylboron (STPB) reagent with intermittent shaking to ensure equilibrium between the chloroform and the aqueous phases. The disappearance of the yellow color in the bottom chloroform layer indicates the endpoint of the titration. The actual concentration of anionic surfactant, such as LAS, is determined by the difference between the amount of the CDBAC added to the sample and the amount of STPB needed for titration.

Calcium, magnesium, copper, lead, and some iron analyses were performed using a Beckman Atomic Adsorption System utilizing a laminar flow burner with an acetylene-air flame and using standard lamps with detection DB-G Spectrometer. Sodium and potassium analyses were run using the same instrument osram lamps.

Total carbon concentrations using 20μl samples were analyzed with the use of the Beckman Infrared Analyzer according to the combustion-infrared method described in Standard Methods (14).

Rhodamine WT was determined using a Turner Fluorometer Model 111 equipped with a Model #110-851UV lamp, a #110-82 primary filter, and a #110-833 secondary filter. During continuous field monitoring the fluorometer was equipped with a flow-thru sampling door and a continuous recorder (Gulton Industries Model 288). For individual rhodamine measurements, the flow-thru door was replaced with a standard cell holder door.

Tritium samples were collected at the site and returned to North Hall for analysis during initial tracer studies. In preparation for analysis, 7 ml of sample were measured into 25 ml polyethylene counting vials equipped with screw caps. After the samples were placed in the vials, 15 ml of Aquasol-2 was added to each vial. Aquasol-2 is a liquid scintillation cocktail prepared by the New England Nuclear Company and is used as a solvent to keep isotopes in solution while counting. A mixture of 15 ml of cocktail with 7 ml of sample was the optimum ratio recommended for Packard Counting Instruments by company representatives. After adding the cocktail, the caps were replaced tightly and the vials were gently shaken to mix the contents. A background sample was prepared using 7 ml of distilled water and 15 ml of Aquasol-2. The samples were then placed in the refrigerator of the scintillation counter described below and three hours were allowed to elapse prior to counting to permit the decay of chemiluminescence and fluorescence. The samples were then counted for 20 minutes each. To obtain the net count per minute (cpm) for each sample, the background gross count was substracted from each sample gross count, and this net count divided by 20.

The Packard Model 3003 Tri-Carb Scintillation Spectrophotrometer used to count samples during initial radiotracer studies is a refrigerated unit with three channels to allow comparison of counts. The machine is a single sample manual unit which required considerable time for sample handling and analysis. This instrument became inoperable on July 31, 1976. All subsequent samples were taken to the RPI Freshwater Institute Laboratory at Lake George and counted on a Beckman LS-133 Liquid Scintillation System. Although the Beckman machine is not a refrigerated unit, it has the advantage of a 100 sample automatic changer plus a digital printout. A tritium standard was prepared to determine the counting efficiency of each liquid scintillation counter. This standard indicated the counting efficiency of the Packard Tri-Carb and Beckman units was 39.4 and 56.4 percent, respectively.

The question arose at the beginning of the project as to whether the samples should be distilled prior to tritium counting. A vacuum distillation system was assembled according to the technique devised by the Radiological Science Laboratory, Division of Laboratories and Research, New York State Department of Health (17). This system was designed to separate tritium from other interferring radionuclides and to remove chemical and/or physical quenching agents. Several samples were counted with and without prior distillation. Comparative results showed very little difference in counts between distilled and undistilled samples; therefore, the distillation procedure was abandoned.

SECTION 6

RESULTS AND DISCUSSION

The study of secondary sewage effluent applications onto sand beds constructed in a natural glacial delta at the Lake George Village Treatment Plant included evaluating both the renovation of wastewater moving through the natural delta sand and the hydraulic characteristics of the sand system. The changes in water quality are presented as those which occurred during vertical flow and those during horizontal flow through the saturated aquifer.

Massive amounts of data have been collected to determine the changes in quality of wastewater moving through the soil. In order to condense the data, seasonal averages were made since the data reflect changes in the temperature as well as the flow regimes within the treatment system. Flows to the treatment plants were significantly influenced by the seasonal tourist influx in the Lake George Area. This major influx occurred from about the middle of June through the middle of September with the peak occurring during July and August. Thus, the summer season represents the greatest sewage loads to the treatment plant. For all monthly data reported herein, the month begins with the 21st of the previous month and continues through the 20th day of the stated month. The data are reported in this way in order that the monthly averages can be converted to seasonal averages without additional significant calculation.

CHANGE IN QUALITY THROUGH THE SOIL

Changes with Vertical Transport

All of the studies of the changes in quality of wastewater during vertical transport through the unsaturated portion of the sand beds were conducted in North sand bed 11. A series of well points was driven at 0.6m (2 ft) spacings to a maximum depth of 4.3 m (14 ft) as shown in Figure 9. The well screens on these wells were located in the unsaturated zone; therefore, samples could only be retrieved from the shallowest well in the series during a flooding period. Only 4 of the 13 lysimeters installed in the unsaturated zone were operable; therefore, all samples retrieved through the unsaturated profile were collected from these lysimeters. Wells 11S and 11D penetrated the saturated aquifer and were used to monitor the quality of the wastewater after mixing with the native ground water.

The average seasonal temperature fluctuations with depth for the year starting in September 1975 through August 1976 are summarized in Figure 11. The temperature of the sewage and applied effluent, as expected, followed the same trends as the ambient air temperatures. The temperatures near the bottom of the unsaturated zone were somewhat more stable, and the temperatures within the saturated aquifer indicated only slight seasonal variations. Thus, during the cooling

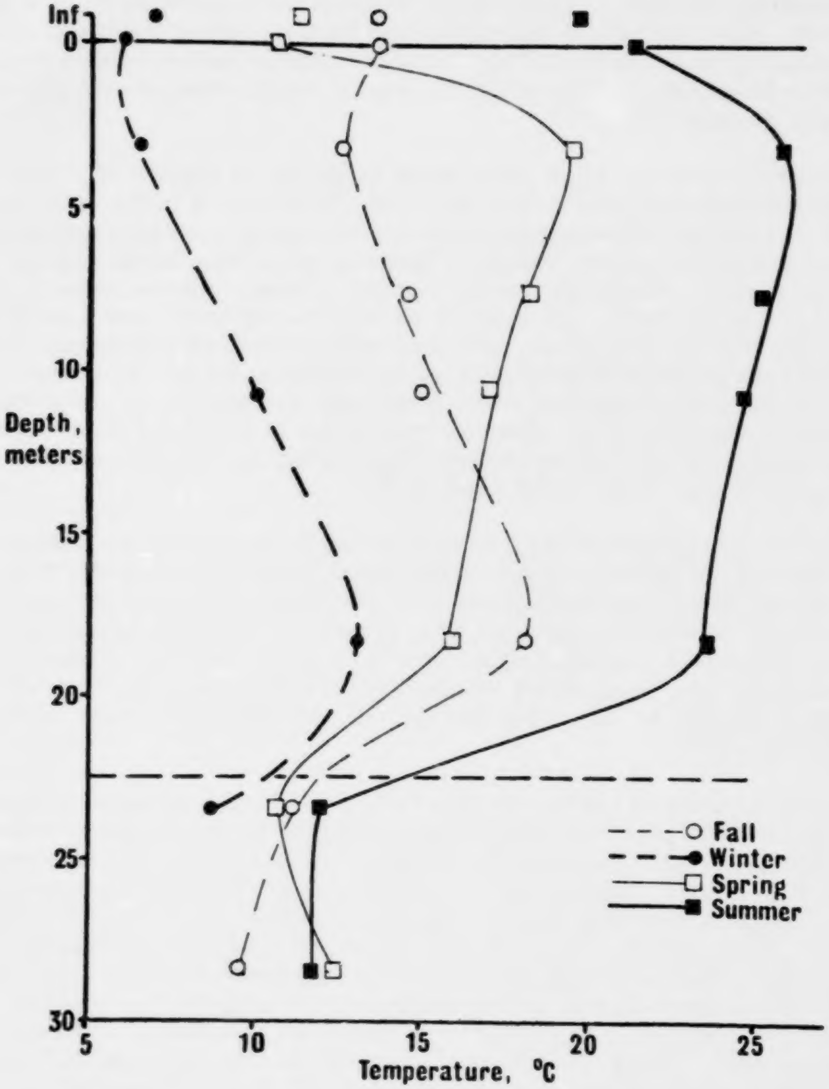


Figure 11. Seasonal Variations in Temperature with Depth in Bed N-11.

seasons of fall and winter, the temperature increased with depth indicating heat stored in the soil profile warmed the infiltrating wastewater. During the warming seasons temperature decreased with depth as the warmer infiltrating wastewater became cooled by soil which had been cooled by water which moved through the profile the previous season. The greatest temperature fluctuation occurred at the 3 m (10 ft) depth which is interpreted as effects of ambient temperature on the percolating effluent. Temperature alone is not suggested to be a pollutant in this system; however, the temperature has been shown to have a dramatic effect on biological systems including the nitrification-denitrification processes. Thus, warmer temperature increasing biological metabolism should enhance purification of sewage (18, 19).

The seasonal variation of pH with depth is shown in Figure 12. The lowest average value determined was 6.5 at the 11 m (36 ft) depth in the fall, and the highest was 7.4 during the summer in the upper part of the saturated zone. The difference in these values indicates there is not a significant change in pH with depth or season. Although the spring and summer curves were similar below the 11 m (36 ft) depth, all seasons except the fall indicated a definite decrease in pH in the top 3 m (9.8 ft) of the sand bed which could be attributed to the production of carbon dioxide produced by biological activity. However, this explanation is not well supported since there was an increase in pH at the 7 m (23 ft) depths in the spring and summer and at the 11 m (23 ft) depth in winter. It should be noted that pH values tended to be higher in the saturated aquifer than in the unsaturated zone of the sand beds.

There was a consistent slight increase in the dissolved solids concentration with depth during all seasons in the unsaturated zone of the aquifer (Figure 13). The highest values were observed to occur during the summer and the lowest during the winter with essentially parallel gradients. Values within the saturated aquifer were consistent throughout the year and were much lower than the values in the unsaturated zone suggesting a considerable dilution of applied sewage effluent high in dissolved solids by the natural ground water which is normally low in dissolved solids.

The dissolved oxygen (DO) content of the percolating effluent is important to any biological process which may be occurring in the sand beds. Maintenance of aerobic conditions will enhance the oxidation of reduced nitrogen compounds to nitrate and generally stabilize other contaminants. The DO concentration is a function of temperature with higher saturation values occurring in colder water. This trend is illustrated by the generally lower values observed during summer and higher values during winter in the unconfined aquifer (Figure 14); however, the highest value observed was in the saturated zone during the summer. As stated earlier, the temperature in the saturated zone remained relatively constant throughout the year. Thus, the DO levels would also be expected to have little variation. There was slightly more variation in DO than anticipated; however, slight aeration of the sample could have occurred due to the sampling technique used. During the fall, winter and spring well 11S was monitored for DO by inserting the DO probe directly into the well. During the summer when the submersible pump was out of service, the DO was measured in samples pumped to the surface. Data show that DO values measured in situ were lower than values measured from the pumped samples, indicating that all the DO values reported

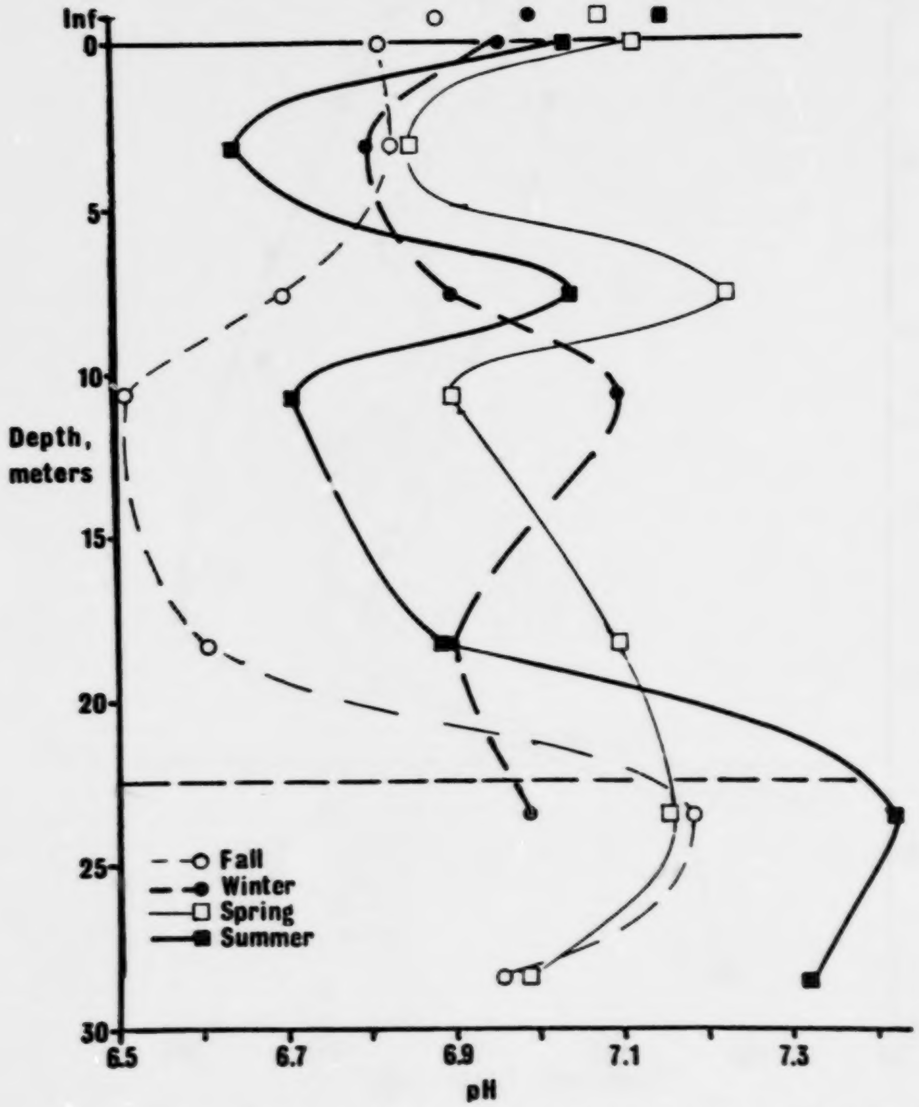


Figure 12. Seasonal Variations in pH with Depth in Bed N-11.

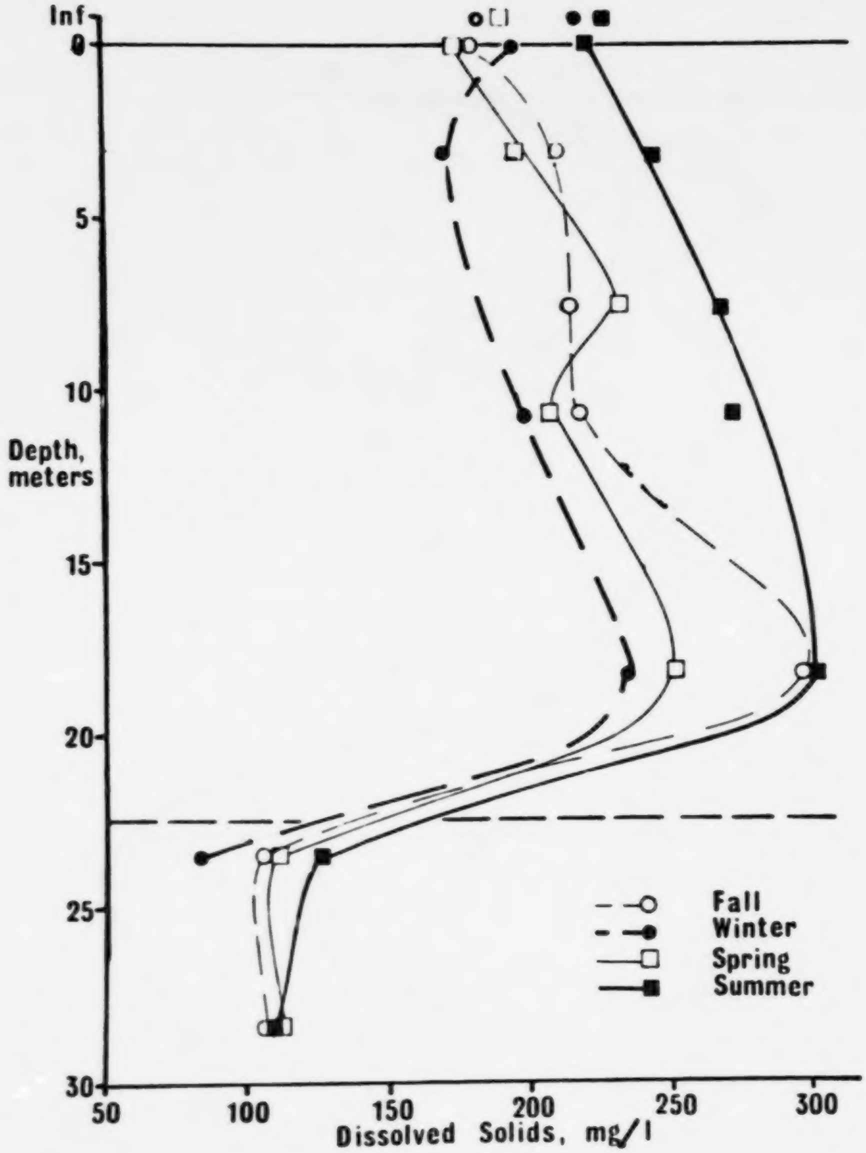


Figure 13. Seasonal Variations in Total Dissolved Solids with Depth in Bed N-11.

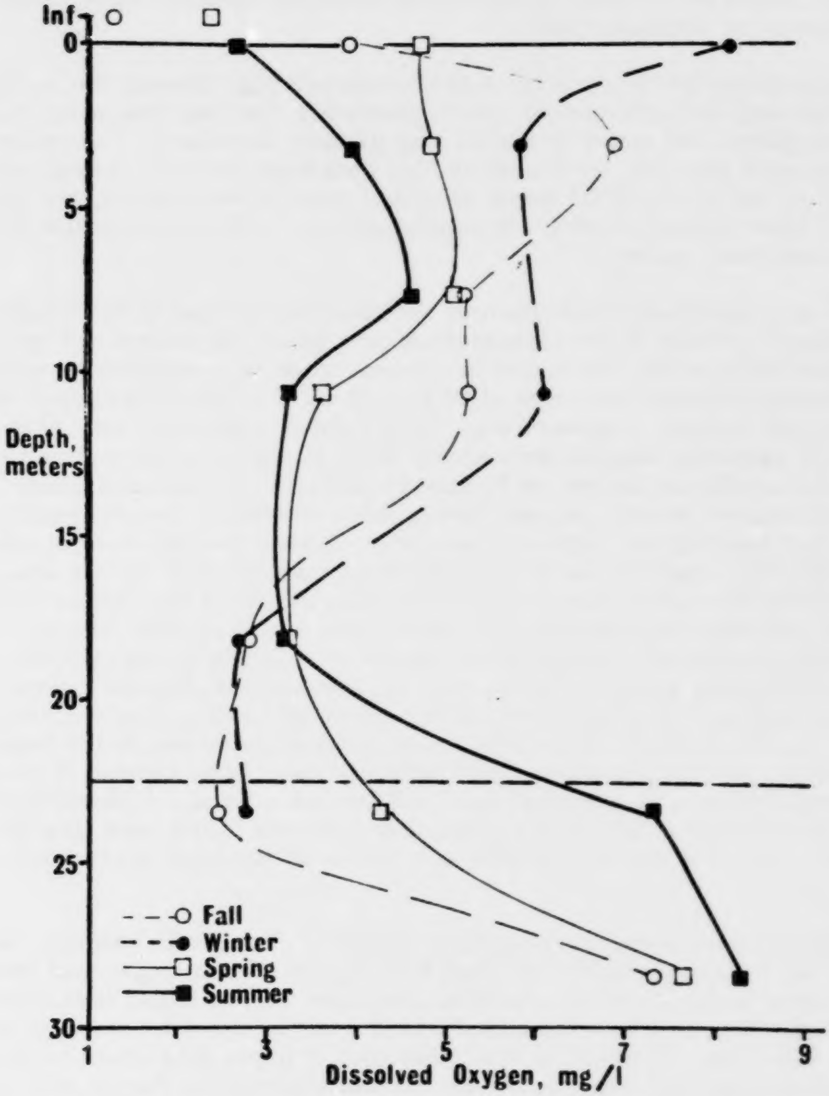


Figure 14. Seasonal Variations in Dissolved Oxygen with Depth in Bed N-11.

in the wells within the sand bed, with the exception of the fall, winter, and spring samples in well 11S, may be slightly high due to aeration during sampling. The DO concentration of the water standing on the surface of the sand beds was definitely influenced by the ambient temperatures. The lowest DO values correlate with the highest temperatures in the summer; while the highest values correlate with the lowest temperature during winter. In general, the DO decreased at the 18 m (60 ft) depth which may be correlated with the denitrification observed immediately above the saturated zone.

The redox potential (Figure 15) was determined only during the spring of 1976 after the sampling instruments were received. During treatment of sewage in the treatment plant, the redox potential was greatly increased. The redox potential in the sand profiles, as illustrated by data from bed N11, remained positive at all but the 18 m (60 ft) depth at which point it became slightly negative. This negative value correlates with the consistently low DO values at the 18 m (60 ft) depth described above.

There was a significant reduction in the chloride content of the wastewater in the conventional portion of the sewage treatment plant, as shown in Figure 16. Through the bed profile, the chloride concentration was essentially constant with depth ranging between extremes of 40 and 80 mg/l in the unsaturated zone for the summer and spring, respectively. In the upper saturated zone of the aquifer seasonal variation ranged from about 40 to 60 mg/l, while values near the bottom of the aquifer as shown on Figure 16 indicate low concentrations in the summer and higher concentrations during fall. However, careful examination of data shows that the high average concentration during the fall was the result of a few exceptionally high values during November while other values observed during this season were quite low, even lower than values at the 11S well depth. These chloride concentration spikes near the bottom of the aquifer during fall cannot be readily explained. The highest values of chloride in the influent raw sewage occurred during winter. These high concentrations correlate directly with the time of salting of highways and subsequent infiltration into the sewage system. The results indicated there was no removal of chlorides in the sand during any season and that chloride could possibly have been used as a tracer of applied sewage effluent in a normal sand infiltration system. Unfortunately, the storage of road deicing salt on the ground in the area of the town garage (Figure 6) prohibited the use of chloride as a tracer in the land application system under study.

Even though there were no consistent trends of changes in calcium versus depth throughout the four seasons, as shown in Figure 17, some general interpretations may be considered. Summer seasonal averages versus depth indicated a general increase with depth in the unsaturated zone followed by a sharp reduction in the saturated aquifer. A possible interpretation of these data could be that calcium is leached from the sand profile by the warm percolate during the summer season then diluted by the lower concentrated ground water in the saturated zone. During winter calcium adsorption to the sand particles is indicated by the very general decrease in concentration of calcium versus depth. It was not possible to correlate the calcium, pH, alkalinity, and orthophosphate concentrations within the sand beds; therefore, calcium should not be considered as a limiting element in the operation of the infiltration system.

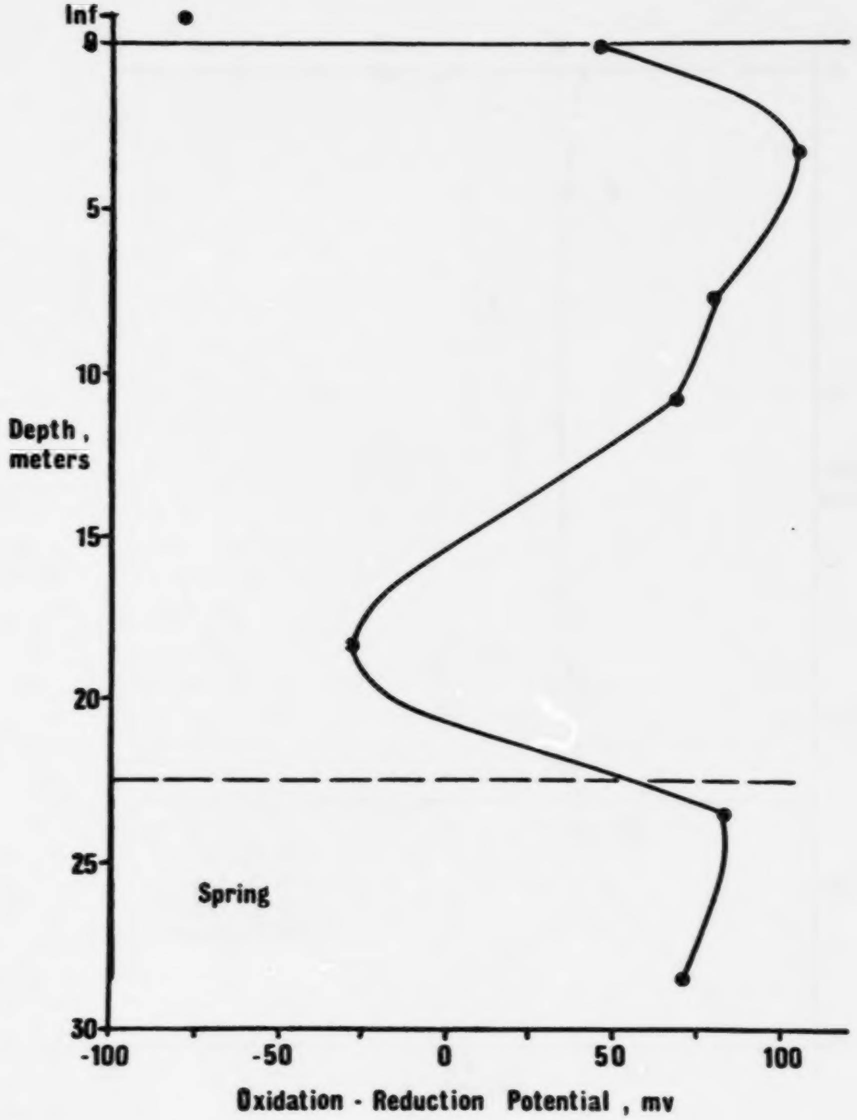


Figure 15. Variation of Oxidation-Reduction Potential with Depth in Bed N-11 During Spring 1976.

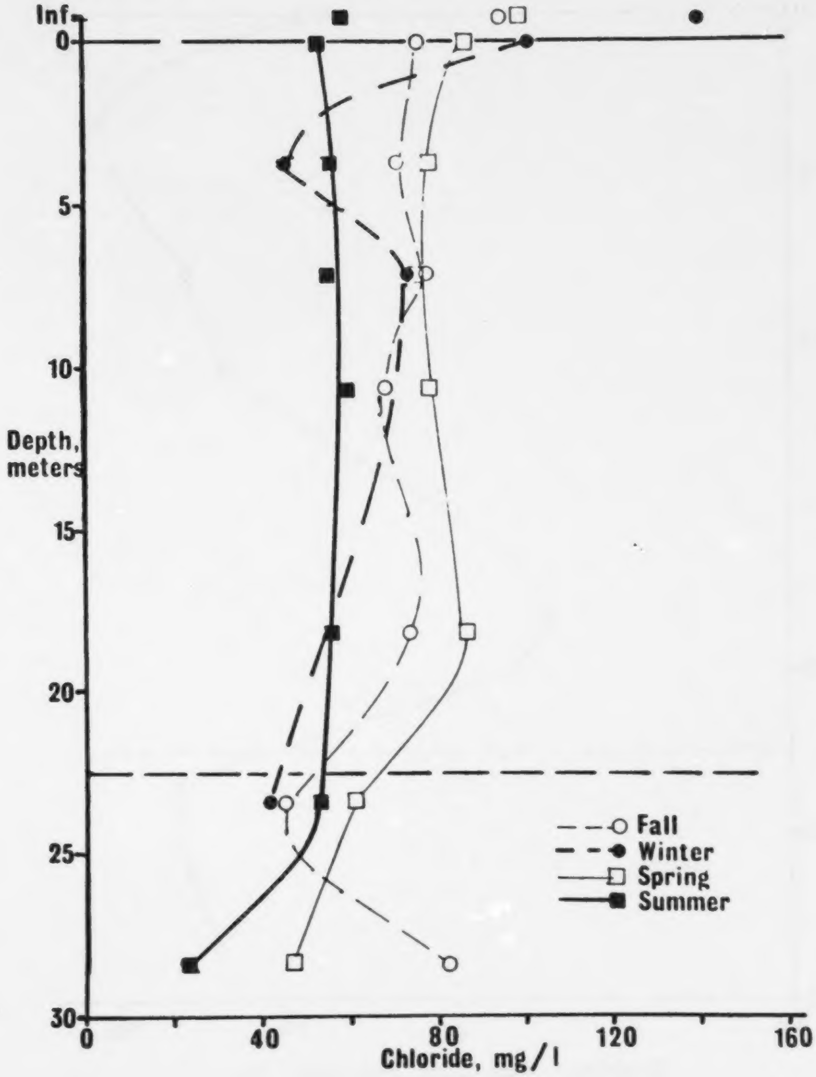


Figure 16. Seasonal Variations of Chloride with Depth in Bed N-11.

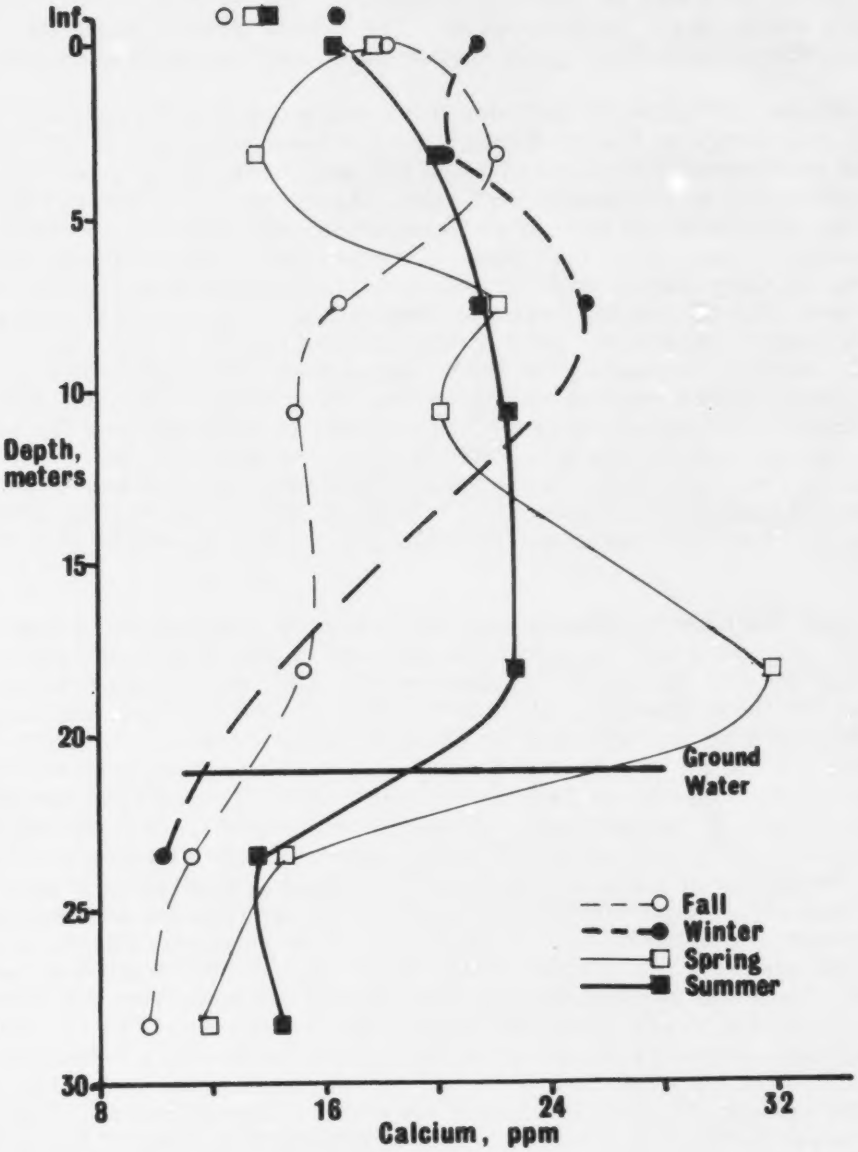


Figure 17. Seasonal Variations in Calcium with Depth in Bed N-11

There was very little significant variation in the magnesium concentration with depth, as shown in Figure 18. The lowest and highest values were approximately 4 and 8 mg/l, respectively. There was a slight trend toward increasing concentrations with depth for all seasons except winter. The interpretation that magnesium in the sand may be leaching through during warmer seasons and stored during winter might be considered. The natural ground water was low in magnesium and diluted the slightly higher concentrations infiltrated from above.

Observation of Figure 19 indicates there was a consistent reduction in alkalinity of the sewage in transit through the treatment plant during all seasons. However, as wastewater infiltrated through the sand beds, there was a significant increase in alkalinity in the unsaturated zone. As the percolate entered the natural ground water, alkalinity, as most other parameters, was reduced by dilution, reaching concentrations lower than those of the influent. The alkalinity measurements correlated very closely with pH and dissolved oxygen data. As the alkalinity increased, the pH and DO decreased indicating the presence of biological activity. Microorganisms in the sand profile utilized the DO in their biochemical mechanisms, thereby decreasing the DO. During their respiration they gave off carbon dioxide which reacted with the water, producing carbonates and hydrogen ions. The increase in carbonate caused an increase in alkalinity and the increase in hydrogen ions decreased the pH. Furthermore, the alkalinity increases were greatest near the surface during the summer, indicating that increased temperature encouraged biological activity. The biological activity during the other seasons was less near the colder surface, but greater at the warmer 18 m (60 ft) depth.

The interrelationship between organic, ammonia, and nitrate nitrogen is discussed by comparing the changes from one form to the other as the percolate moved through the soil profile during the summer, fall, and spring seasons. In order to compare these changes, all three forms measured (nitrate, ammonia, and total Kjeldahl nitrogen) are shown in one figure for each season. Insufficient data were obtained during the winter to accurately support interpretations of changes for this season, therefore these data are not presented. The data for summer and fall, Figures 20 and 21, respectively, showed similar trends. In both seasons there was a decrease in ammonia and Kjeldahl nitrogen with a corresponding increase in the nitrate nitrogen at the 3 m (10 ft) depth. At slightly greater depths, there was a reduction in nitrate with a significant increase in the ammonia and Kjeldahl nitrogen, followed by reduction in both these parameters at the 18 m (60 ft) depth. In both seasons, the nitrate content at the 18 m (60 ft) depth was less than 1 mg/l; thus, there was apparently some loss of total nitrogen from the infiltrating water in the form of nitrogen which escaped to the atmosphere. The only nitrogen form of significant concentration found in the pumped wells was a relatively high nitrate concentration which occurred in well 11S during the summer. During the spring the results (Figure 22) showed somewhat different trends. There was a substantial increase in the nitrate concentrations in the upper 3 m (10ft) of the sand beds with a subsequent gradual reduction with depth to less than 1 mg/l at the 18 m (60 ft) depth. At the 3 m (10 ft) depth the ammonia and total Kjeldahl nitrogen were reduced to less than 1 mg/l followed by a uniform concentration with depth down to the top of the saturated zone. The results of the spring data indicated that there was an initial oxidation of the ammonia and total Kjeldahl nitrogen to nitrate with a possible subsequent reduction of the nitrate directly to nitrogen gas. It is also interesting to compare the nitrate content during the

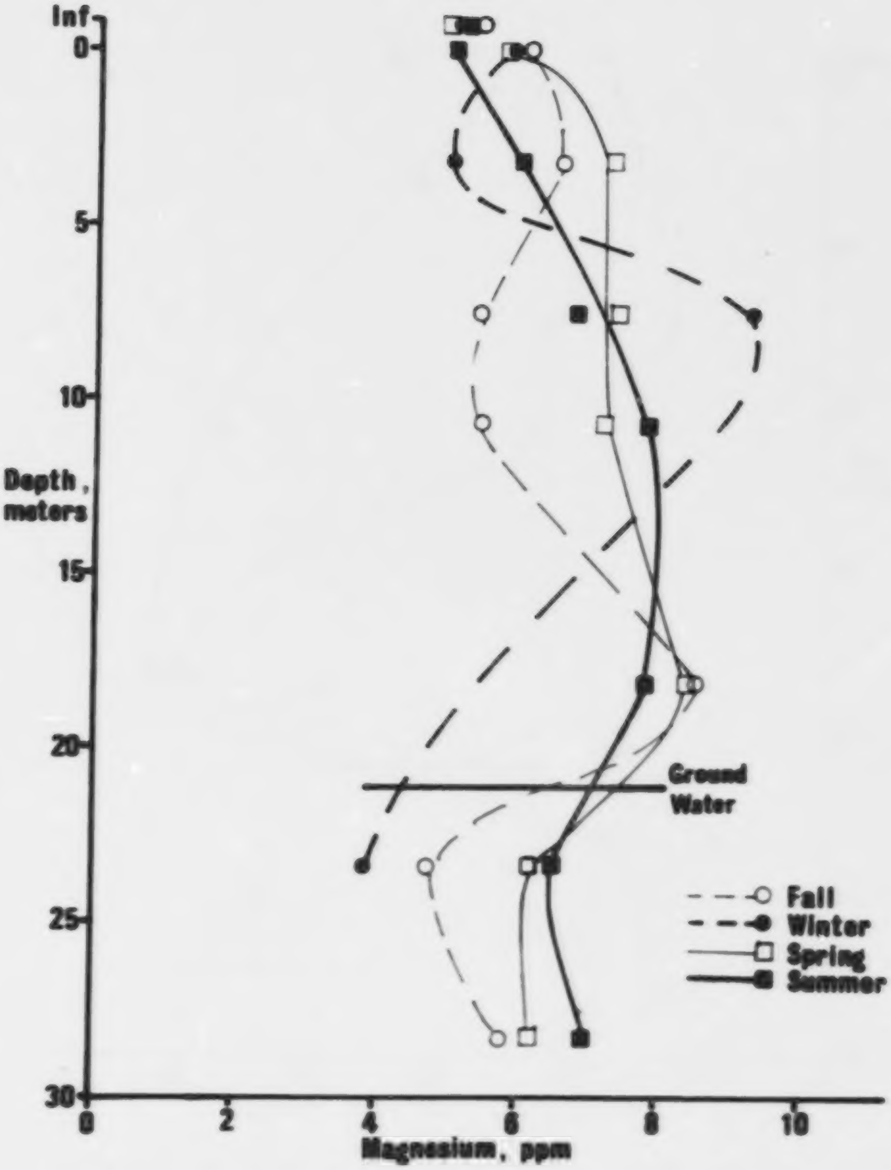


Figure 18. Seasonal Variations in Magnesium with Depth in Bed N-11.

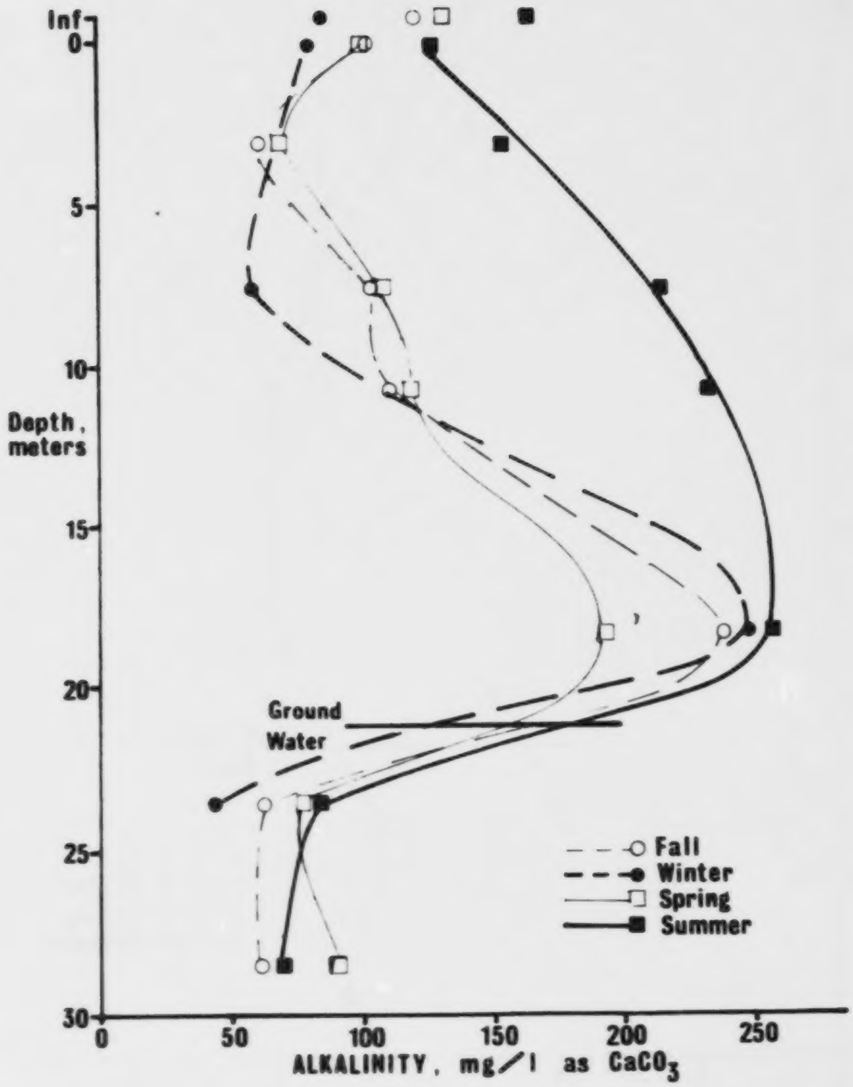


Figure 19. Seasonal Variations in Alkalinity with Depth in Bed N-11.

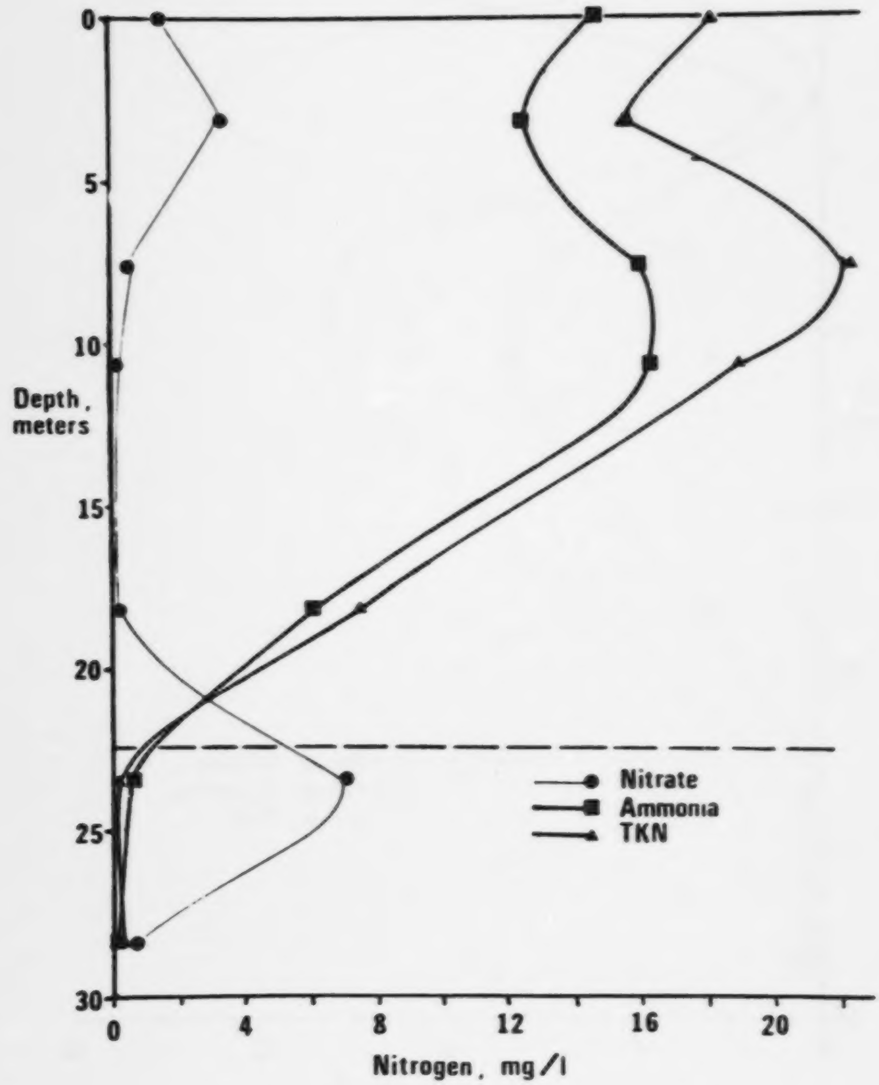


Figure 20. Variations in the Forms of Nitrogen Measured with Depth in Bed N-11 During the Summer of 1975.

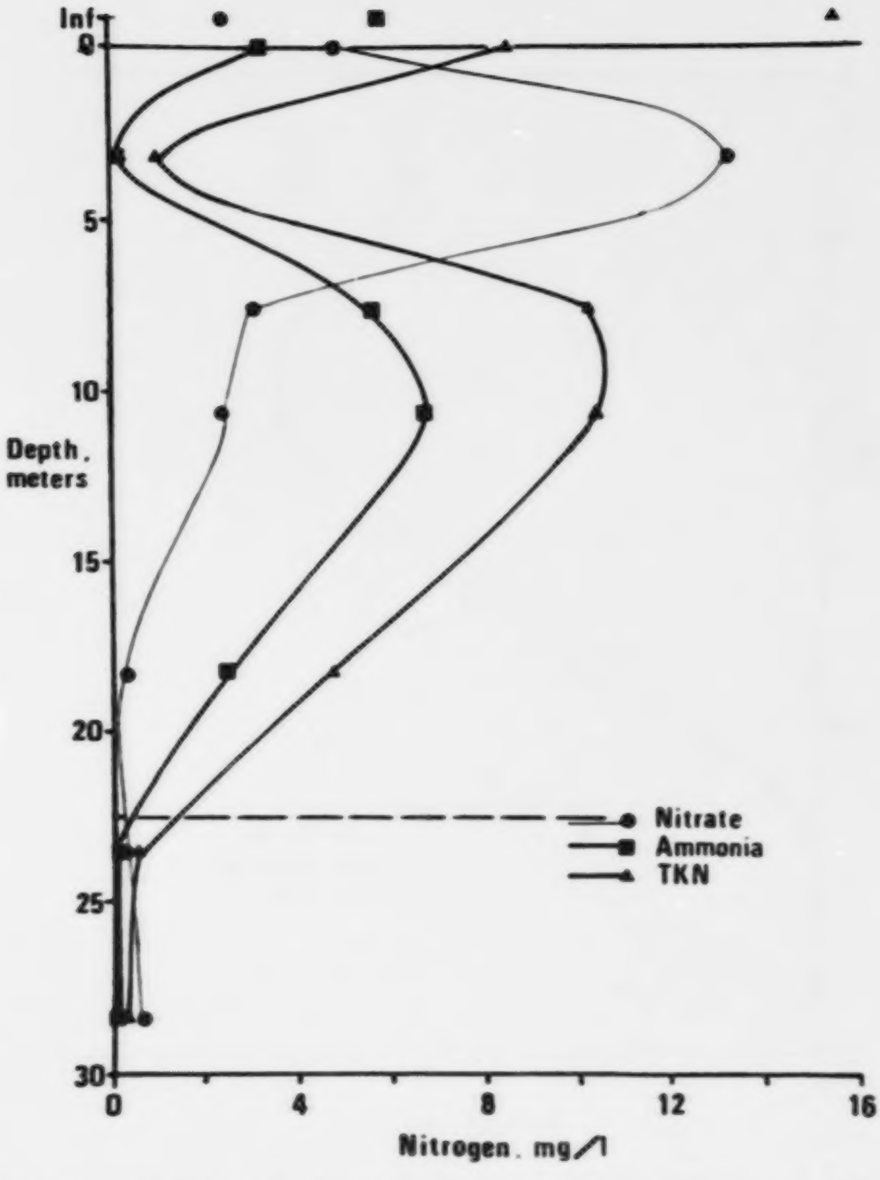


Figure 21. Variation of Various Forms of Nitrogen Measured with Depth in Bed N-11 During the Fall 1975.

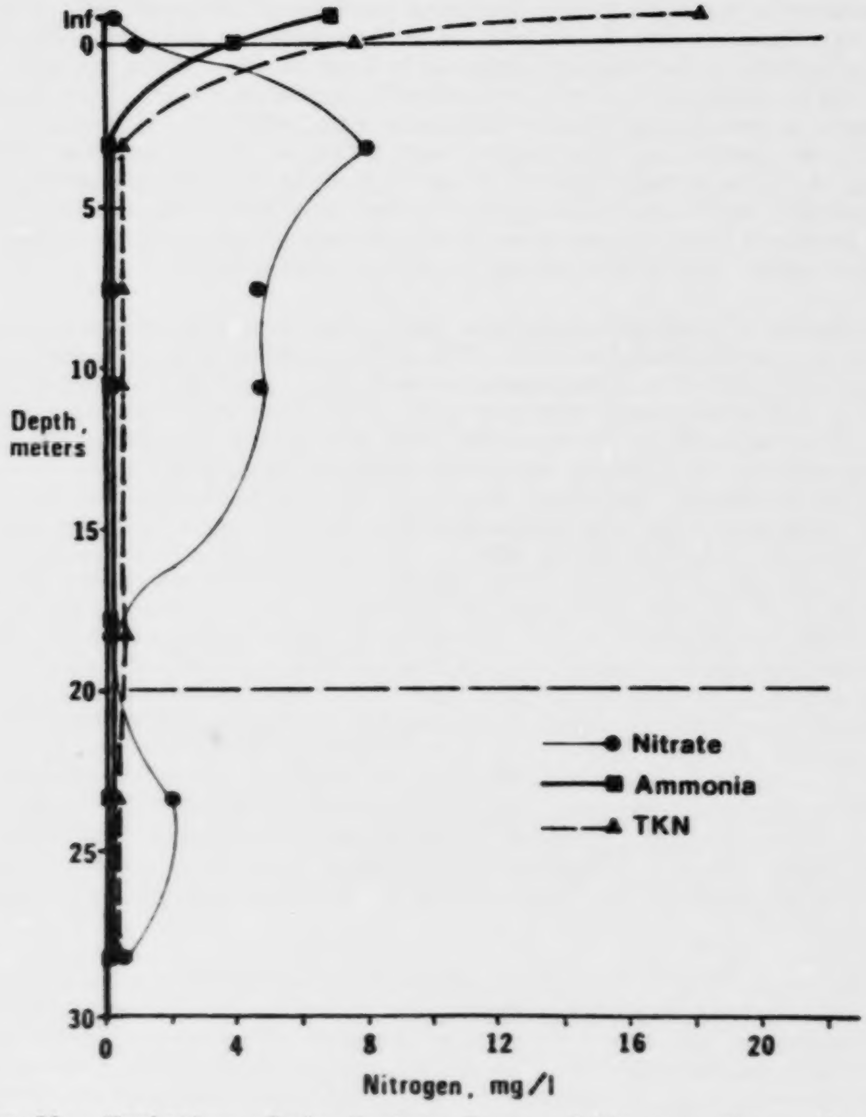


Figure 22. Variation of the Various Forms of Nitrogen Measured with Depth in Bed N-11 During Spring 1976.

spring with the oxidation reduction potential measured during the same period as shown in Figure 15. The two curves are nearly identical in shape, with the possible exception of slightly higher values of redox potential in the saturated portion of the aquifer. This correlation supports the theory that oxidation to nitrate occurred in the upper 3 m (10 ft) of the sand bed with subsequent reduction of the nitrate to nitrogen gas in the lower portions of the sand bed. One question which these results do not resolve is the relatively high concentration (7 mg N/l) of nitrate in the seepage adjacent to West Brook. Since the total nitrogen at the 18 m (60 ft) depth in bed N-11 was usually less than 7 mg/l, it is obvious that the nitrate in the seepage did not originate from bed N-11. However, several of the sand beds, particularly the newer south beds, are much less than 20 m (65 ft) deep. A bed with only 5 m (15 ft) of sand could contribute considerably more total nitrogen to the saturated ground water zone than deeper beds. This may be the source of high nitrate water found at some locations in the upper portion of the aquifer and in the seepage adjacent to West Brook.

The two forms of phosphorus studied were orthophosphate and total phosphate. Total phosphate includes both ortho and the polyphosphate forms which generally are hydrolyzed to orthophosphate with time. Biological systems require phosphorus as a nutrient along with nitrogen. The ortho and total phosphate phosphorus are compared for the summer, fall, and spring in Figures 23, 24, and 25, respectively. Insufficient data were collected during the winter to accurately evaluate winter treatment; therefore, the winter results are not presented. In all cases the orthophosphate was reduced to less than 0.1 mg P/l by the time the effluent reached the 10 m (35 ft) depth. The total phosphorus was also reduced significantly in the top 3 m (10 ft) in the summer and fall but was followed by an increase in concentration at the 8 m (25 ft) depth. During the spring the concentration of total phosphorus was very similar to that of orthophosphate. Slight amounts of total phosphorus were observed in the shallow pumped well during the summer and fall, but during the spring in the deeper pumped well, the levels were consistently less than 0.1 mg/l. During the spring, a significant reduction of orthophosphate occurred in the wastewater while in transit through the sewage treatment plant, but during the fall, there was a slight increase in orthophosphate, possibly indicating a conversion of polyphosphate to orthophosphate in the treatment system. In all instances, by the time the percolating wastewater reached the 10 m (35 ft) depth, the orthophosphate was reduced to levels lower than could be achieved by conventional physical-chemical treatment methods for phosphate removal.

There are many factors that affect soluble iron concentration, including pH and biological activity. High concentrations of iron in ground water indicate anaerobic conditions, since in the presence of dissolved oxygen, iron will be precipitated and remain insoluble. Therefore, a high soluble iron content is indicative of low pH, high carbon dioxide concentrations, and high bacterial activity. In addition, increasing iron concentrations would be expected to increase alkalinity because the reaction converting iron to a soluble reduced form also increases bicarbonate as shown in the following reaction:

$$\mathrm{FeCO_3 + CO_2 + H_2O \leftrightarrow Fe^{+2} + 2HCO_3^-}$$

Data collected for iron analysis versus depth were averaged for the various seasons and presented in Figure 26. Low concentrations of iron were found in the raw

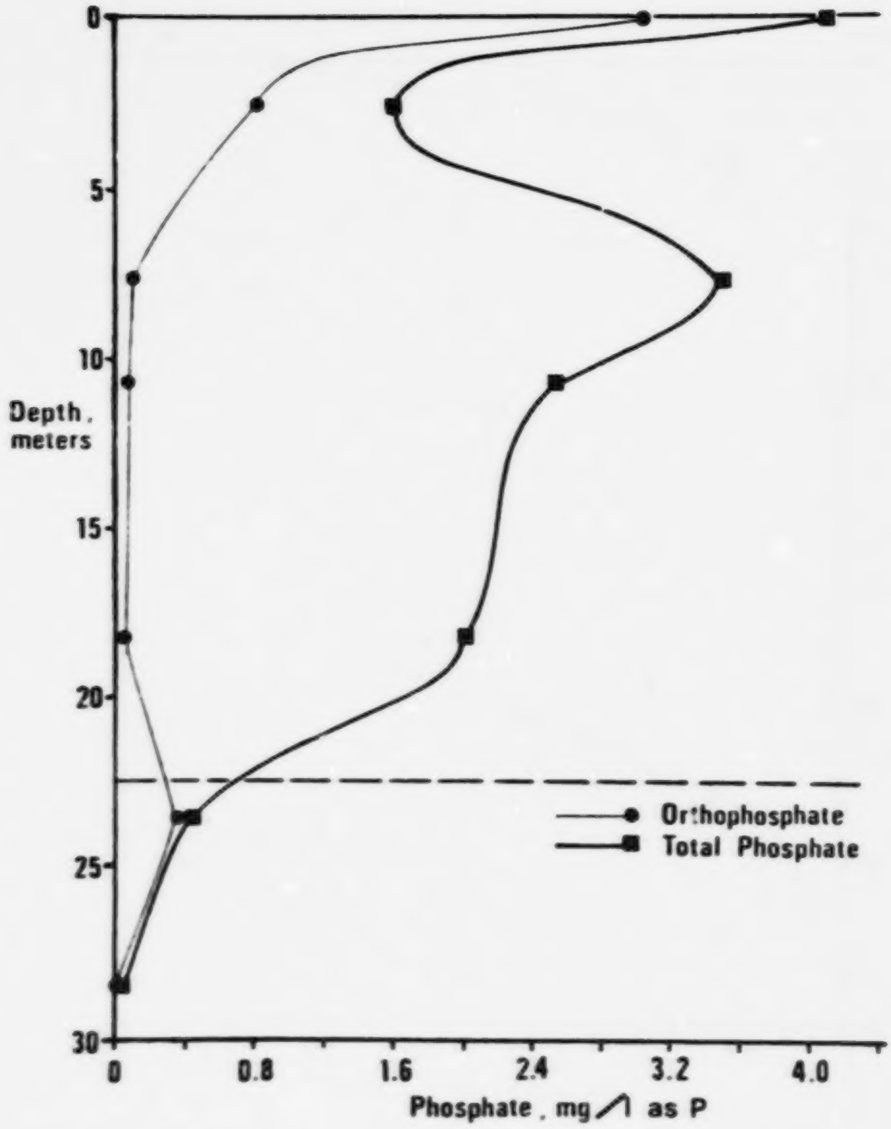


Figure 23. Variation in Ortho and Total Phosphate with Depth in Bed N-11 During Summer 1975.

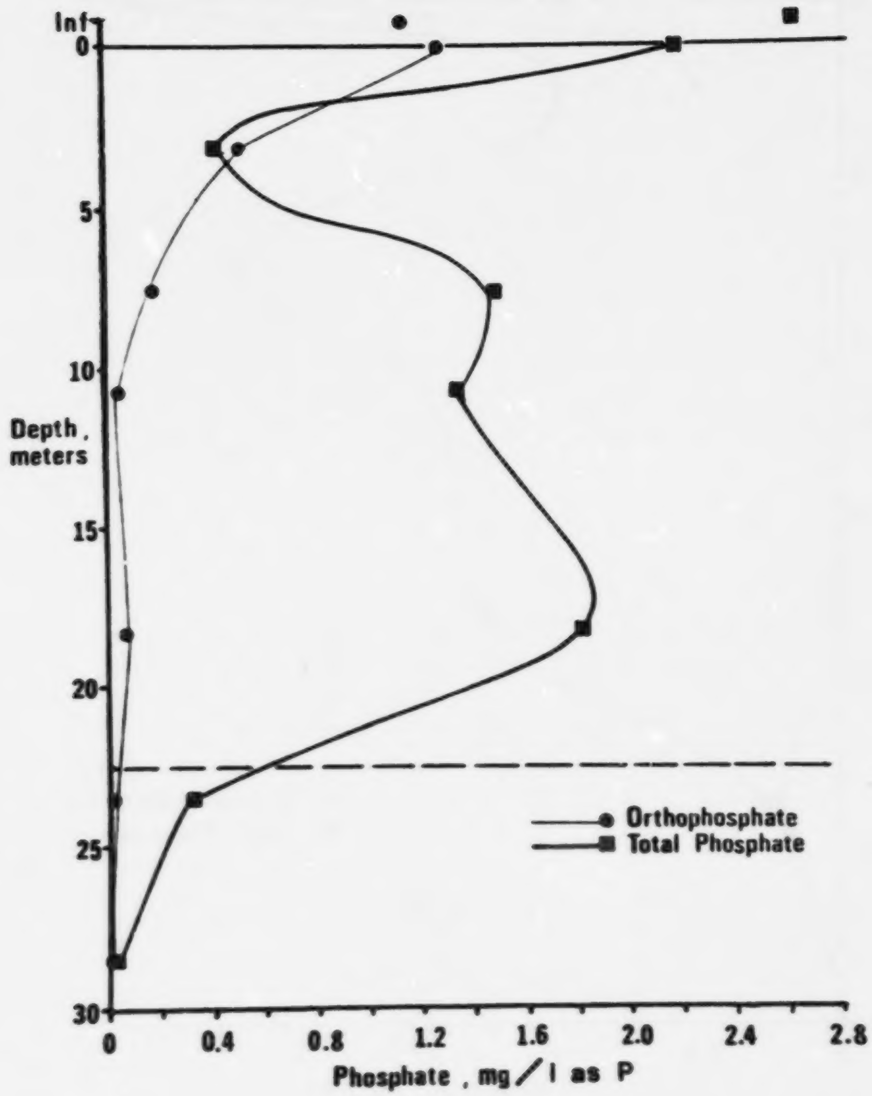


Figure 24. Variation in Ortho and Total Phosphate with Depth in Bed N-11 During Fall 1975.

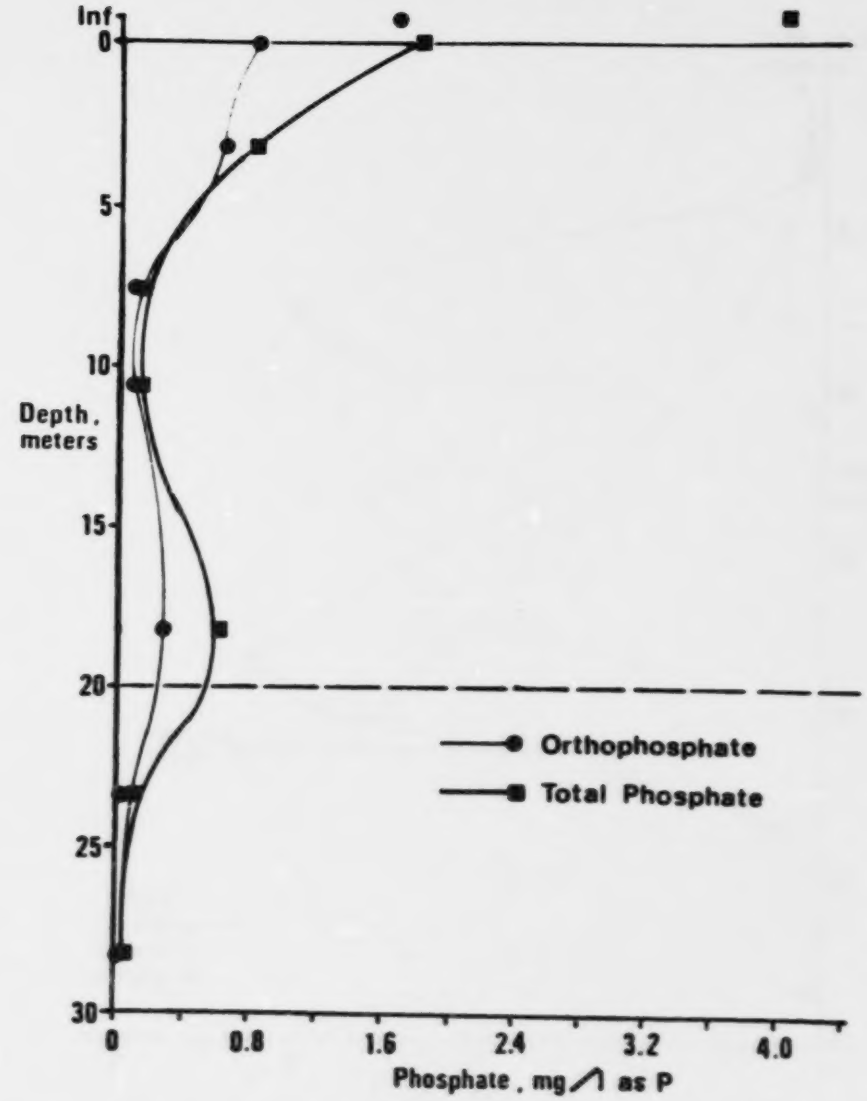


Figure 25. Variation in Ortho and Total Phosphate with Depth in Bed N-11 for Spring 1976.

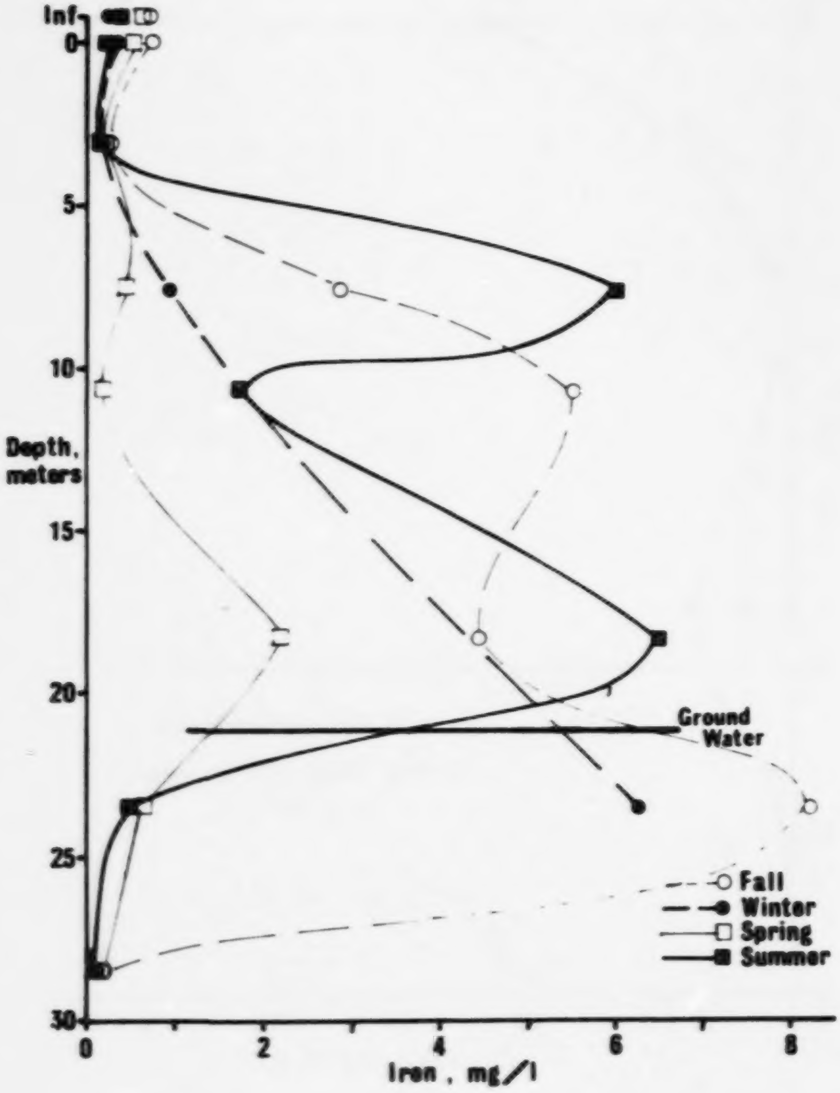


Figure 26. Seasonal Variations in Iron with Depth in Bed N-11.

sewage and applied effluent. There was a general trend toward increasing concentrations of iron with depth in the unsaturated zone, with the highest increase observed during the summer and fall. During the spring and summer the iron concentrations were low in the saturated aquifer; however, during the fall and winter, high values were observed in well 11S. In evaluating the data, it must be pointed out that during the fall, winter, and part of the spring the pump in well 11S was inoperable and samples were collected using a bailer. The well was cased with steel pipe, and without the use of the submersible pump, it was impossible to collect representative samples free of rust. Thus, the low values observed in all the other wells and the low values measured most other times in well 11S were probably more realistic values. There was a definite correlation between the iron concentration, redox potential, dissolved oxygen, and alkalinity. There did not appear to be any correlation between iron concentration and phosphate removal. It is possible that iron was solubilized in the upper portion of the sand beds where it immediately precipitated the phosphate, and then any further solubilization of iron remained as an increase in iron content since there was no phosphorus remaining to precipitate the iron.

Specific measurements for sodium and potassium were made in order to provide some background data for use of these elements in the tracer studies. Sodium and potassium, being quite soluble, will follow the movement of water through the soil. Sodium is found in some concentration in almost all natural waters, but potassium is generally present only in waters containing human sewage effluents; thus, by determining the potassium to sodium ratio, the presence of sewage effluent can be detected. This was particularly necessary in this study since ground water contamination from road deicing salts is known to occur near Lake George. Previous studies have shown that the road deicing salt used in this area contains no potassium (20). Figure 27 shows all values of sodium to be within 10 to 18 mg/l with no significant changes with depth or season. Lower values were found in the saturated portion of the aquifer indicating dilution of the sewage effluent by the native ground water.

The potassium concentration of the applied effluent from the sewage treatment plant also varied only slightly (4 to 10 mg/l) as shown in Figure 28. There appeared to be a very slight increase in concentration with depth; however, dilution in the saturated aquifer was again illustrated with potassium concentrations less than 1 mg/l. This evaluation confirms that the source of potassium is from municipal wastewater, and potassium would be expected to be found in low concentrations in uncontaminated water. The ratio of potassium to sodium is shown in Figure 29. It may be seen that the ratio generally exceeded 0.4 in areas where there was contamination from municipal wastewater and was in the range of 0.1 in the relatively uncontaminated ground water confirming the use of the ratio as an indicator of the presence of human waste discharges.

Measurements were made for copper in all of the samples; however, all of the concentrations were less than the lowest detectable limit on the Atomic Absorption Spectrophometer System used in the study (0.05 mg/l). With concentrations this low, it was impossible to evaluate any seasonal changes with depth. These concentrations were less than the drinking water standard (21).

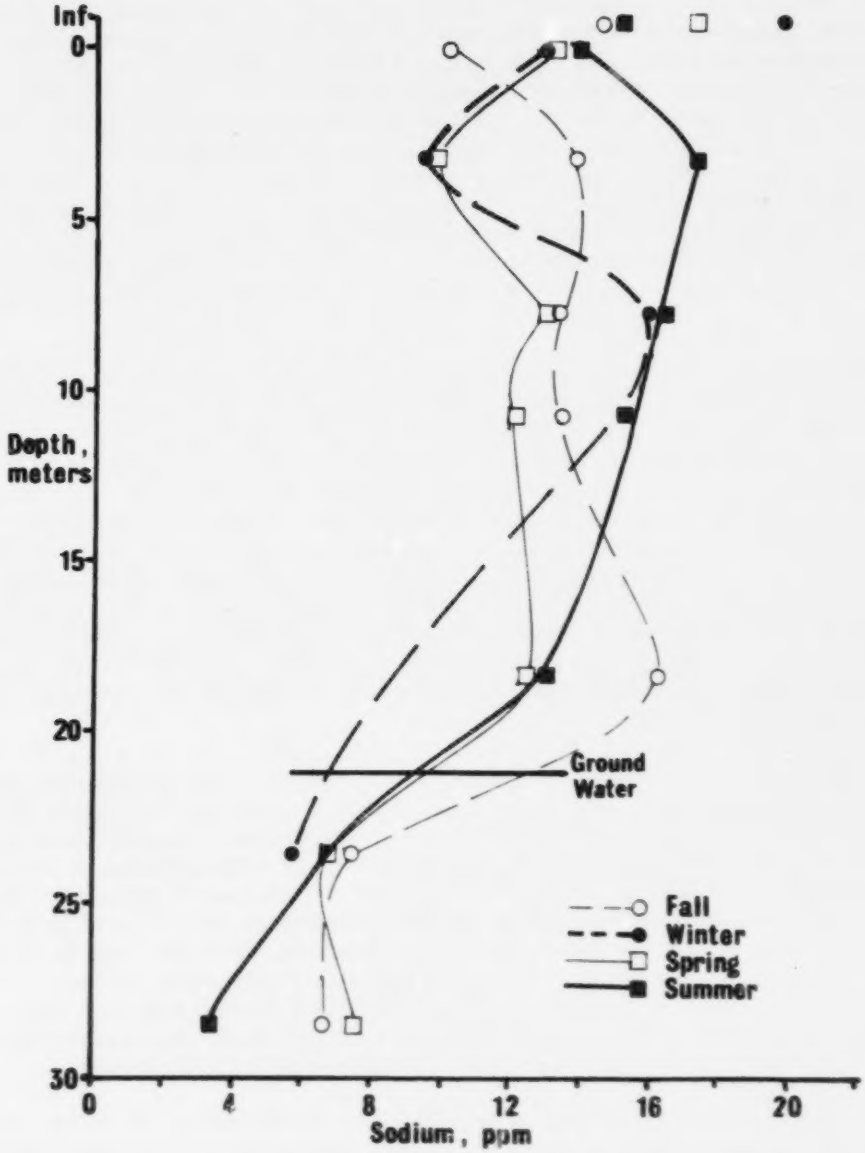


Figure 27. Seasonal Variations in Sodium with Depth in Bed N-11.

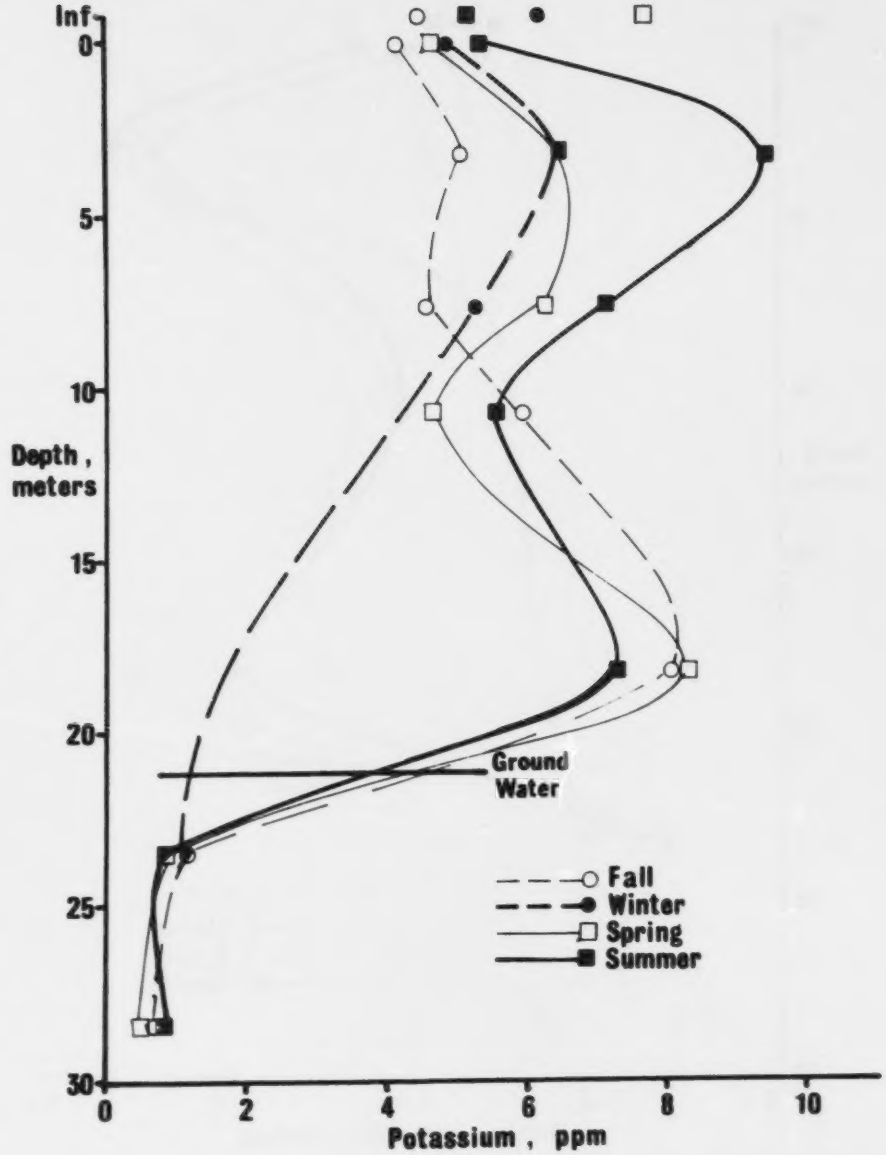


Figure 28. Seasonal Variations in Potassium with Depth in Bed N-11.

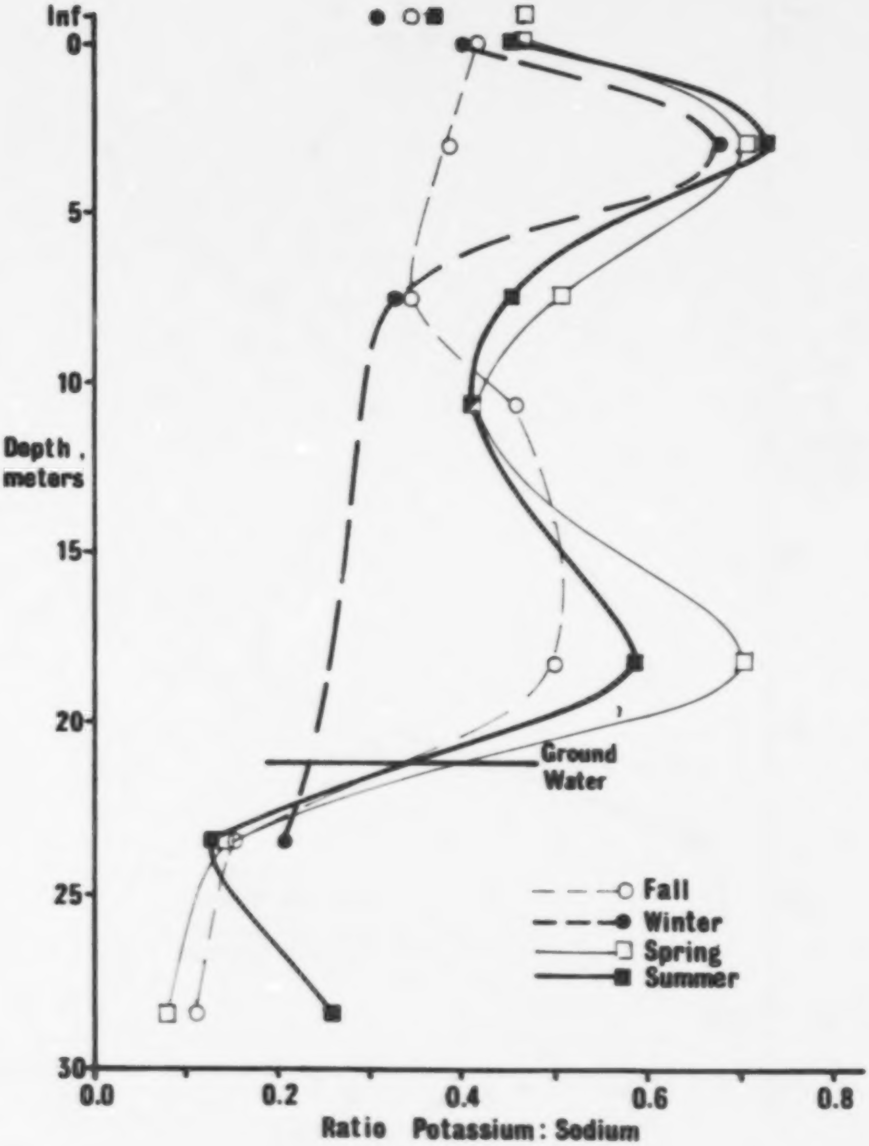


Figure 29. Seasonal Variations in the Ratio of Potassium to Sodium with Depth in Bed N-11.

Changes With Horizontal Transport

After the applied sewage flows vertically through the sand beds, it enters the ground water aquifer below and then flows in a northerly direction toward the seepage at West Brook approximately 600 m (2,000 ft) away. The studies of change in quality with distance within the saturated aquifer were conducted by two different researchers. The studies from November 1974 through September 1975 were conducted by Hajas (22) and the study was continued by Chiaro (23) from September 1975 through August 1976. Hajas's results include all of the data obtained prior to the initiation of this project, including studies by the NYSDEC.

The data collected at each sampling location were condensed for interpretation by averaging for each of the four seasons. The averaged seasonal data are displayed in a series of bar graphs in Figures 30 through 41. The order of seasons in the bar graphs in Figure 30 through 39 is spring, summer, fall, and winter. These graphs are based on data from the spring of 1976, the summers of 1975 and 1976, the fall of 1975, and winter of 1976. Figures 40 and 41 show the seasonal variations in chlorides and dissolved solids for all samples collected through September 1975. In these figures, the seasons are listed in the order of fall, winter, spring, and summer.

The observation wells are identified according to the numbers as indicated on Figure 6. The letter following the well number indicates the relative depth of the well point into the aquifer at locations where more than one point was installed. These letters are AA, A, B, C, and D indicating the order of shallowest to the deepest, respectively. Wells 11 are pumped wells and are labeled 11S and 11D to indicate shallow and deep pumped wells, respectively. The West Brook surface flow sampling sites upstream and downstream of the seepage streams are designated as stations WBUS and WBDS, respectively. The stations monitoring the seepage above Gage Road and below Gage Road are noted as SA and SB while the treatment plant influent and effluent are designated as INF and EFF, respectively. Well elevations, water depth, and top of rock elevations are compared in Figure 7.

Figures 30A, 30B, and 30C show the seasonal variations in temperature for each sampling point. As expected, the surface waters and the shallowest wells (3A, 4, 5, 7, 10) showed the largest fluctuations in temperature while the deeper wells (2, 3) were relatively stable. The sewage plant effluent being applied to the infiltration beds was warmer in the summer (19.3°C) and colder in the winter (6.0°C) than the raw sewage influent (18.6°C in summer and 6.9°C in winter), reflecting the effect of the ambient air temperature on the sewage as it passed through the treatment plant. The temperature of the effluent in the winter remained warm enough to melt the ice cover on the sand beds and thaw the soil surface, allowing infiltration to continue through the subfreezing winter season. Well 9 had temperature fluctuations about one season behind those of surface waters and other wells. The highest temperature in well 9 occurred in the fall (15.26°C) and the lowest in the spring (9.54°C). One explanation of this phenomenon was that the lateral flow from the sand beds to well 9 is several months duration, causing delay in temperature fluctuations between the infiltration beds and well 9. The cluster of wells at the well 6 location also exhibited similar temperature lay characteristics, although not as dramatic. Later, tracer studies discredited this explanation.

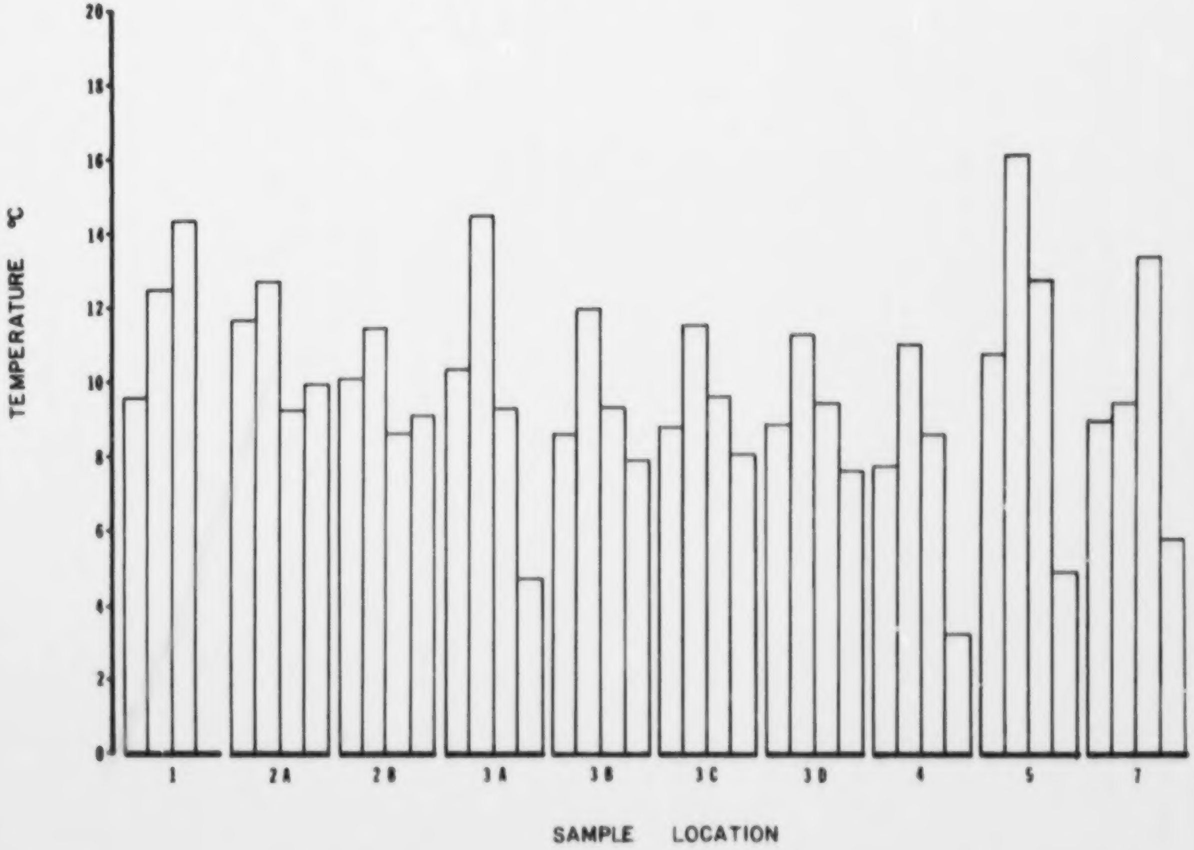


Figure 30A. Seasonal Variations in Temperature. The Bars Represent, in Order, Spring, Summer, Fall and Winter.

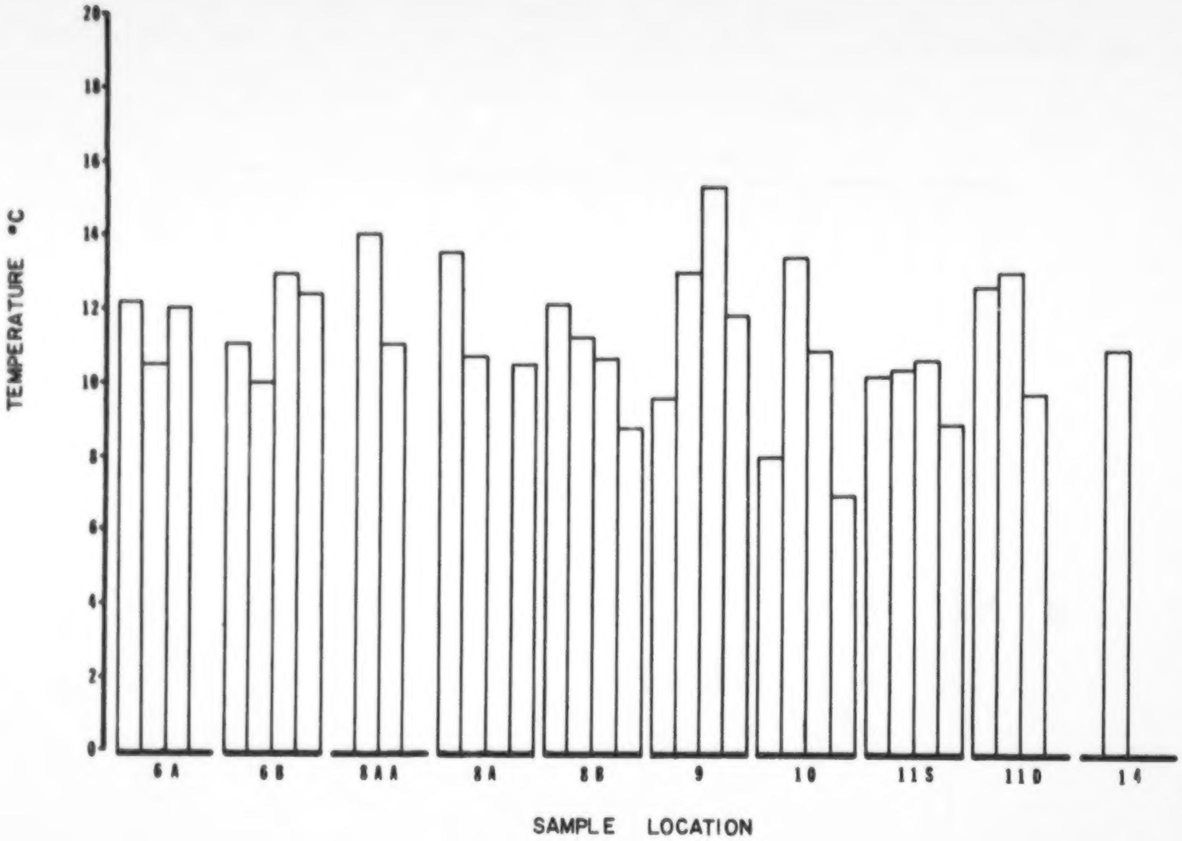


Figure 30B. Seasonal Variations in Temperature. The Bars Represent, in Order, Spring, Summer, Fall and Winter.

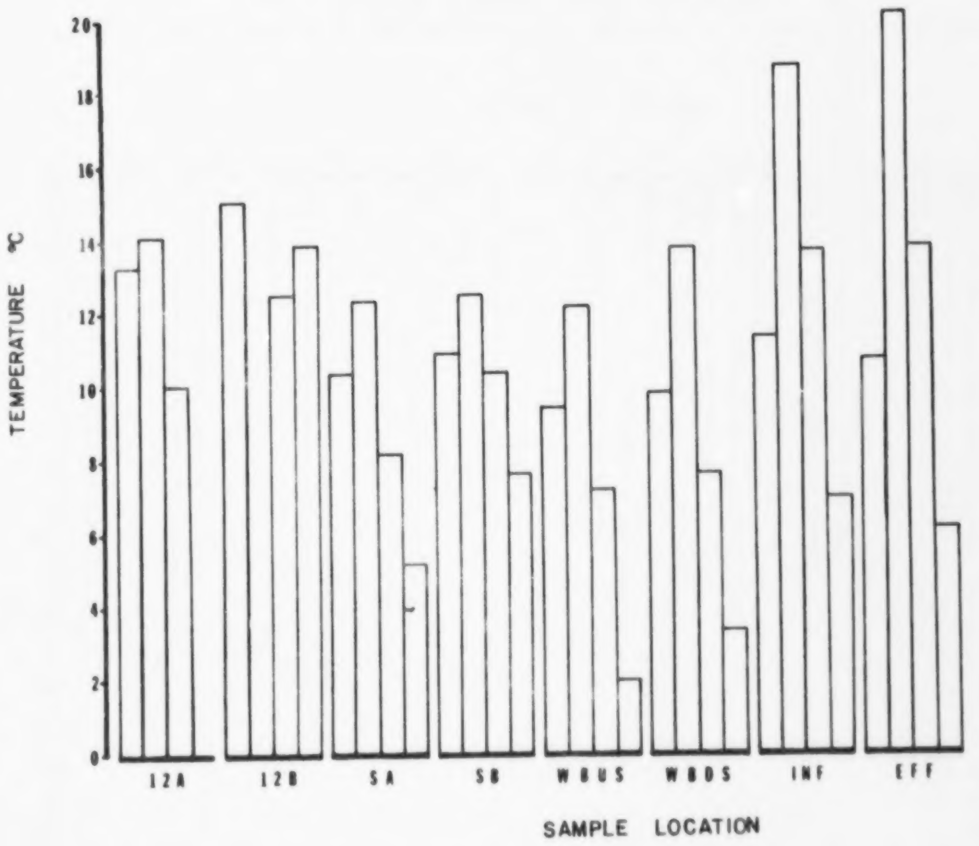


Figure 30C. Seasonal Variations in Temperature. The Bars Represent, in Order, Spring, Summer, Fall and Winter.

The influence of the seepage stream temperature on West Brook was also apparent. West Brook, downstream of the confluence of the seepage inflows, was from 0.5°C to 1.6°C warmer in winter than upstream because of the warmer inflowing sewage streams.

Dissolved Oxygen (DO) concentration in water is a function of water temperature; therefore, the seasonal DO changes can be directly correlated with seasonal temperature changes. Figures 31A, 31B, and 31C illustrate the seasonal DO changes at each water sampling station. In almost all cases as would be anticipated, the DO was higher in the winter and lower in the summer. West Brook and the seepages had the highest DO concentrations of all stations measured. DO levels at WBUS were always above 8.0 mg/l and station SB always remained above 5.9 mg/l even during the summer. These high concentrations might be expected because these shallow, rapidly flowing streams tend to reoxygenate quickly. The DO of the treatment plant effluent was about 2.5 mg/l higher than that of the influent due to the reduction of the BOD load and as a result of oxygenation received when spraying the sewage over the trickling filters. The control wells (4, 7, 10) had relatively low DO concentrations, particularly in the summer months when DO levels fell below 1 mg/l in both wells 4 and 10. Wells 2A, 6A, 12A, and to some extent, 8AA had higher DO concentrations than respective deeper wells in these individual well clusters indicating the upper portion of the saturated zone is more oxygenated by oxygen rich percolate. Wells 9, 6B, 2B, and 3D had DO concentrations which were generally less than 1 mg/l. Wells 1 and 5 also exhibited depressed DO levels which were probably the result of BOD from the effluent. Wells 3A and 4 had low DO concentrations which is attributed to biological decomposition in the swampy area of these locations preventing reaeration below the soil surface. The DO data collected for wells 11 cannot be considered reliable since samples were secured by pumping, and data indicates air was entrained during the sampling process.

The redox potential of most wells was in the range of +50 to +150 mv as shown in Figures 32A, 32B, and 32C. These values indicate the presence of oxidative conditions. The summer redox value for well 11D was zero, indicating low DO which further confirms the probable error in DO measurements for this well and implies a similar error for well 11S as discussed above. The only other low redox potentials were the summer values for the influent and effluent which is normal since warm weather decreases the saturation concentration of oxygen in water. The DO measured in the influent to the plant probably represented oxygen that became entrained as the raw sewage splashed into the Parshall flumes leading to the primary settling basins. The oxygen concentration was further increased in passing through the trickling filters, and the amount of oxidizable organic matter in the water was reduced as the sewage passed through the treatment system. All these factors account for the rise in redox potential of the sewage water from the time it entered the treatment plant until it was discharged onto the sand beds.

Seasonal fluctuations of pH at each sampling point are shown in Figures 33A, 33B, and 33C. Two processes which are accepted as influencing pH are the production of organic acids and carbon dioxide by soil bacteria and, possibly, the conversion of ammonia to nitrates, both of which lower the pH. The influent and effluent generally remained stable around pH 7.0. In wells 1 and 5 the pH

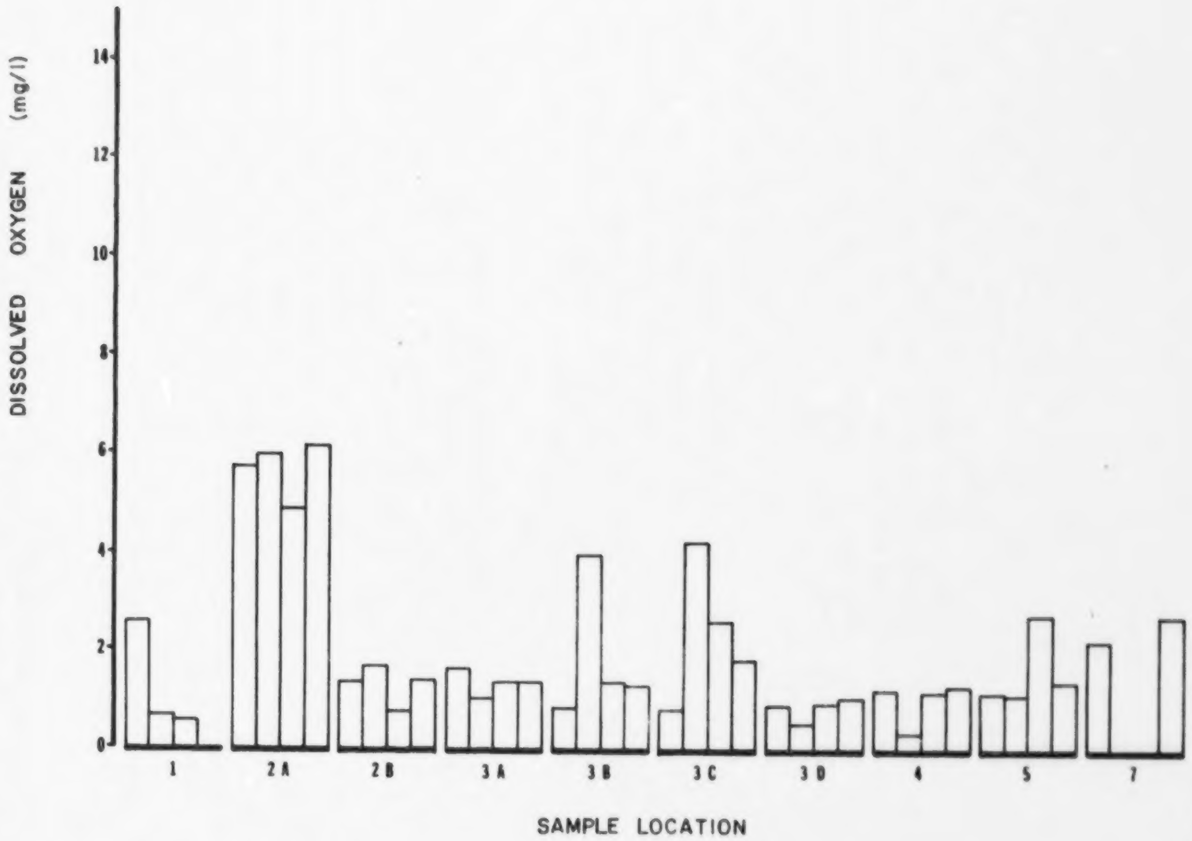


Figure 31A. Seasonal Variations in Dissolved Oxygen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

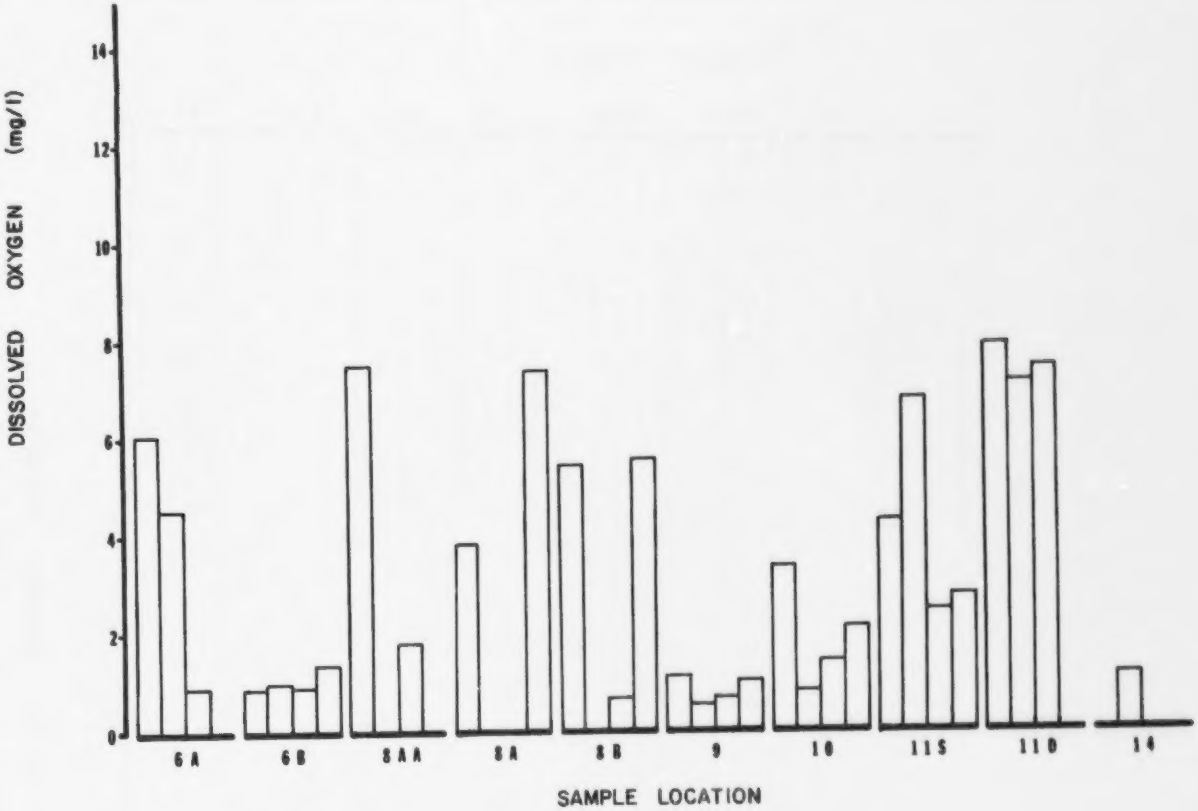


Figure 31B. Seasonal Variations in Dissolved Oxygen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

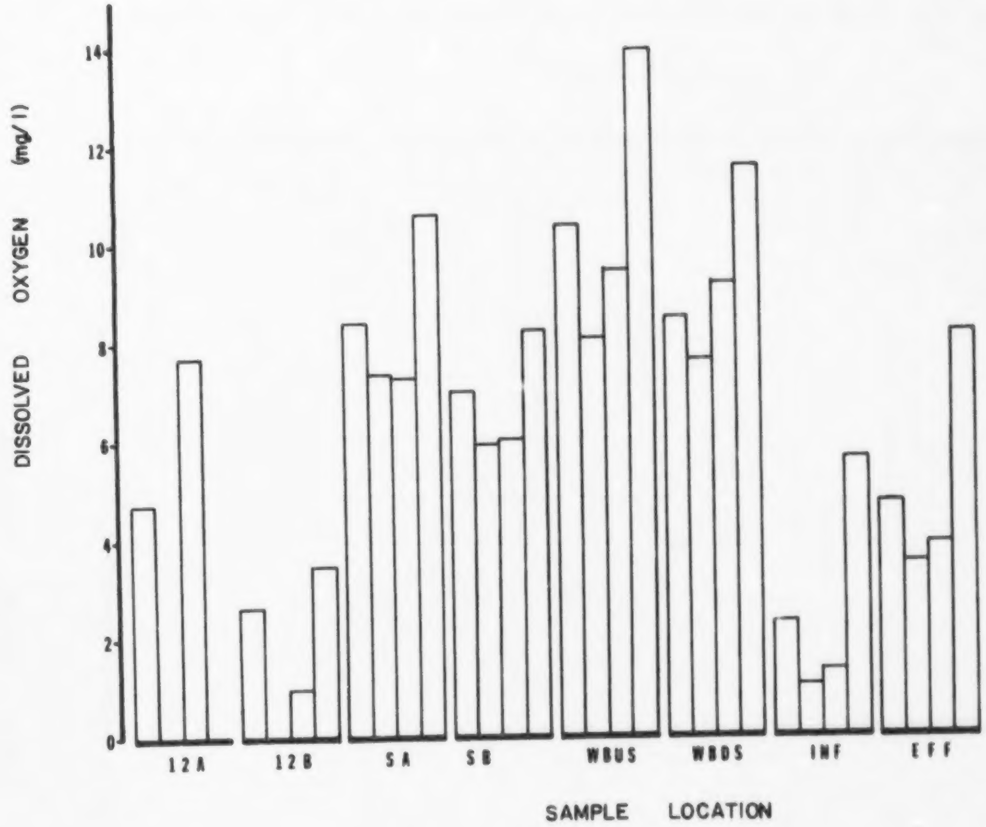


Figure 31C. Seasonal Variations in Dissolved Oxygen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

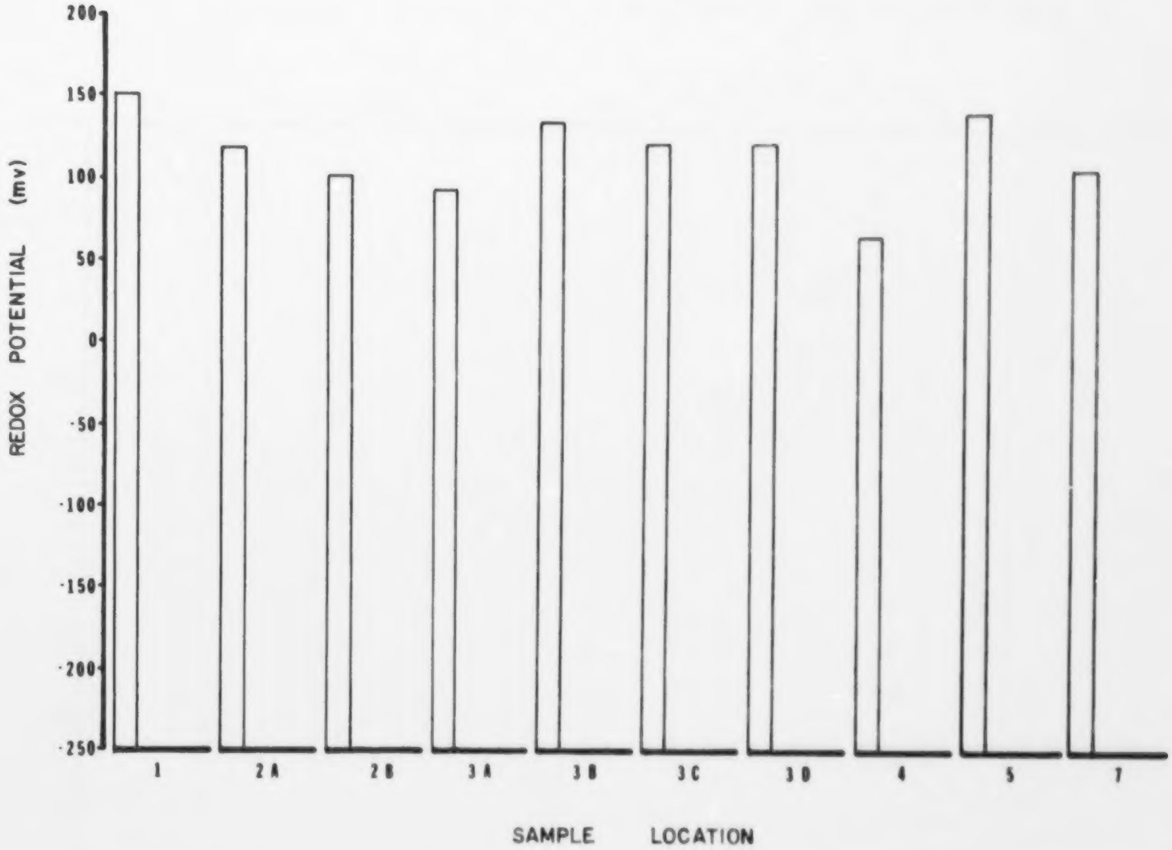


Figure 32A. Variation in Redox Potential During Spring 1976.

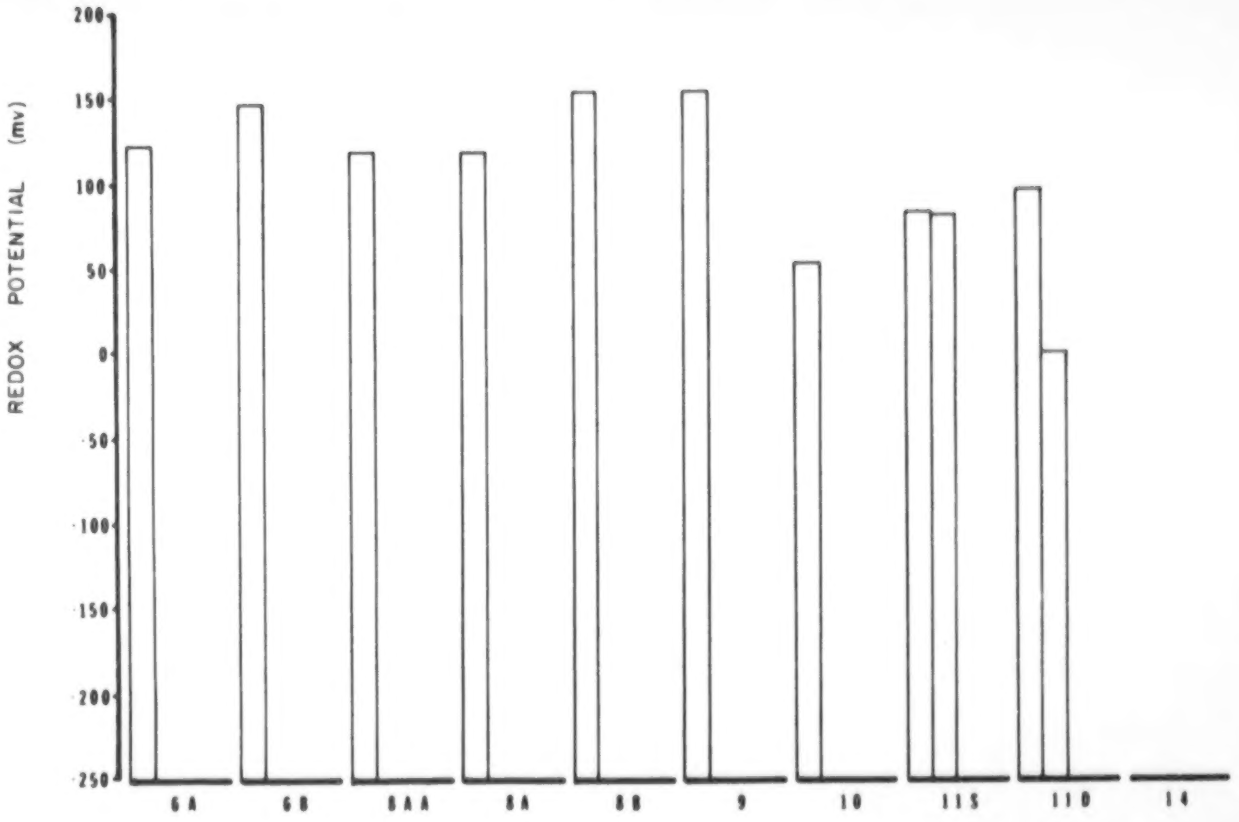


Figure 32B. Variation in Redox Potential During Spring 1976.
(With some Results for Summer).

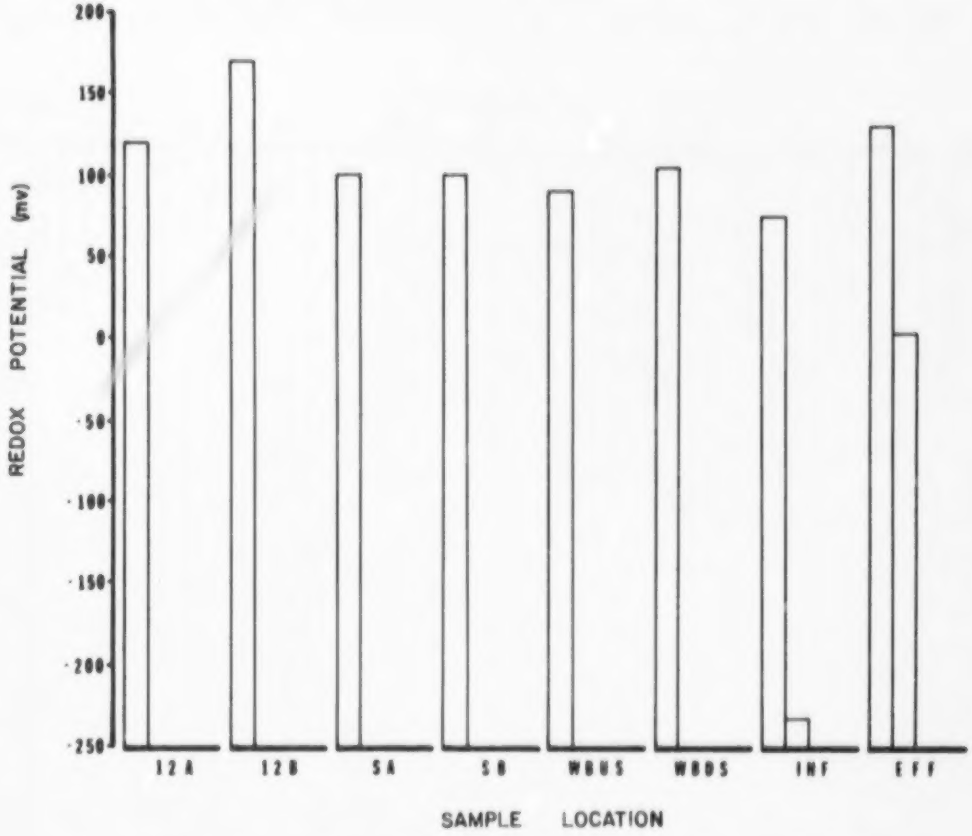


Figure 32C. Variation in Redox Potential During Spring 1976. (With some Results for Summer).

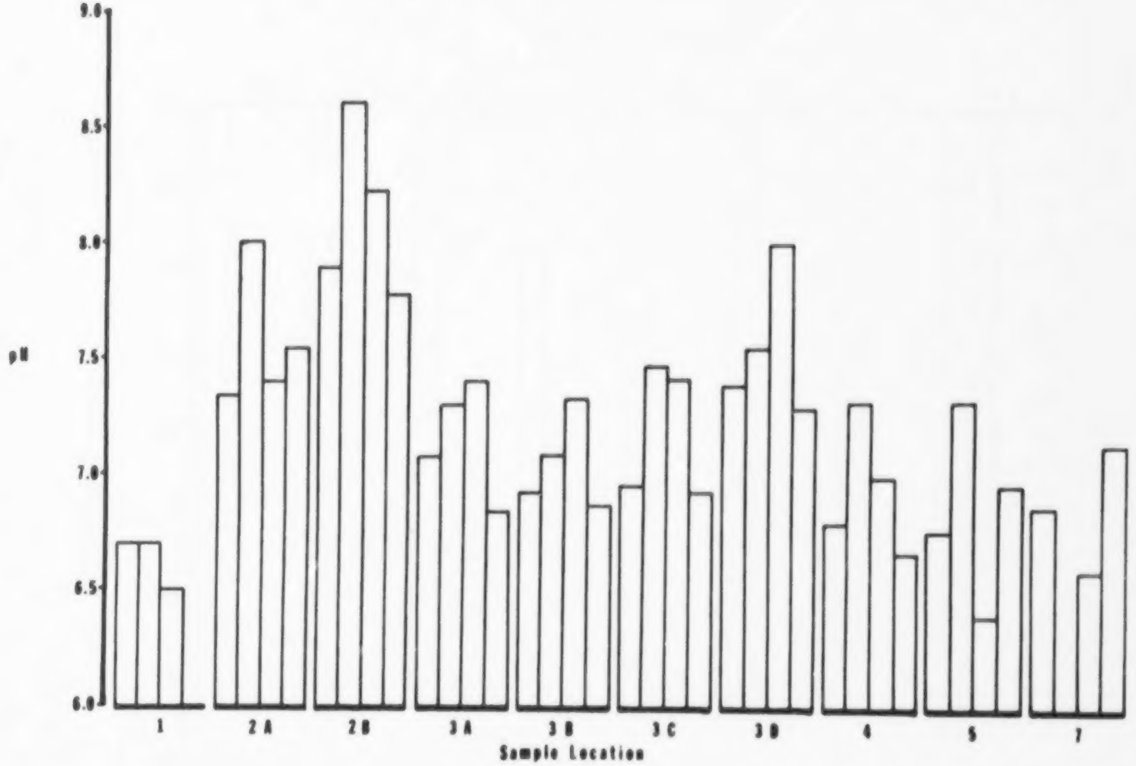


Figure 33A. Seasonal Variations in pH. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

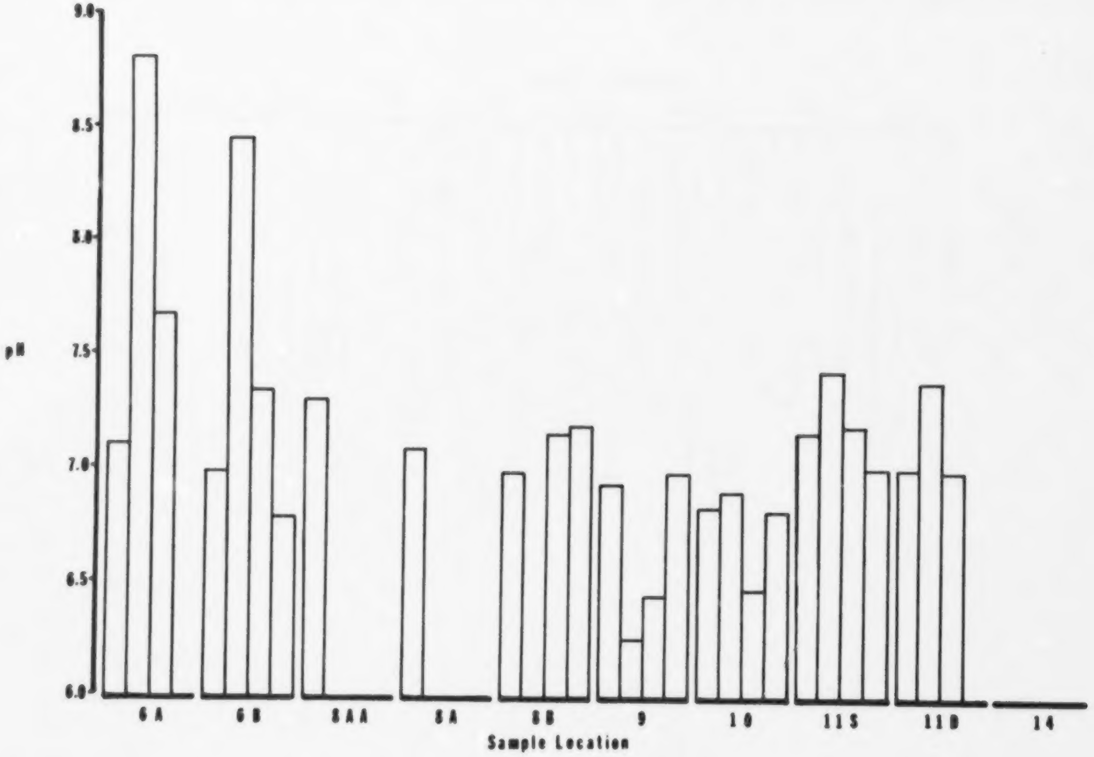


Figure 33B. Seasonal Variations in pH. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

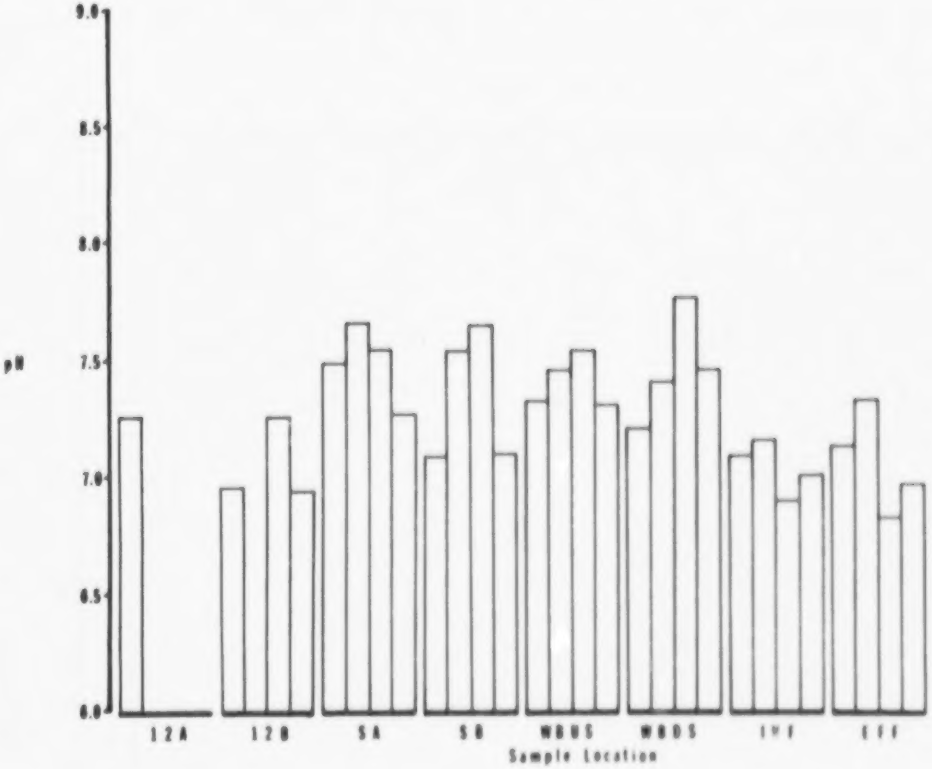


Figure 33C. Seasonal Variations in pH. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

was somewhat below neutral and ranged from 6.5 to 6.7. Most other wells exhibited a rise in pH which was very pronounced in wells 6A, 6B, 2B, and 3D but not as pronounced in wells 11S and D. This increase may have been the result of the reduction of nitrate to ammonia (or nitrogen gas) under anaerobic conditions. Increases in pH were most evident in the summer when DO concentrations were low, and sewage flows were high. Wells 8, 9, and 12 did not experience large pH increases, indicating aerobic conditions were maintained at these wells. Wells 3A, 3B, and 3C, the seepage streams, and West Brook all had similar pH patterns indicating that the sewage flowed deep in the aquifer at well 3D and that wells 3A, 3B, and 3C were influenced by local runoff. The sewage effluent deeper in the ground water could be entering Lake George where the aquifer emerges in the lake beneath the lake surface. The pH of the natural ground water was slightly less than 7.0, as measured in control wells 4, 7, and 10.

Log scales were used to construct bar graphs to illustrate seasonal averages of concentrations of various nitrogen forms due to their wide variation in concentration. The seasonal averages for nitrates are shown in Figures 34A, 34B, and 34C, while ammonia data appear in Figures 35A, 35B, and 35C, followed by Figures 36A, 36B, and 36C which contain results of the total Kjeldahl nitrogen determinations. These data are discussed together since they are all interrelated.

Well 7, a control well which was theoretically uncontaminated, had relatively high nitrate levels, especially in the fall when the nitrate level reached 2.20 mg/l. The ammonia nitrogen concentrations in the well were less than 0.10 mg/l, but in the fall total Kjeldahl nitrogen (TKN) concentration rose to 1.17 mg/l. Well 7 is located in a small depression uphill from the treatment plant. It was learned that plant personnel had been dumping the scrapings from the sand beds in the area of this depression. Apparently runoff from the dumped material drained into the depression, infiltrated into the aquifer, and contaminated well 7 destroying its usefulness as a control well. Well 10, which was also designated as a control well, had nitrate levels in the range of 1 mg/l in both the summer and fall. Ammonia levels reached as high as 0.40 mg/l in the fall, and TKN levels were above 1.0 mg/l every season, indicating well 10 may also have been a poor selection for a control well. The sewage may have flowed far enough east to affect the ground water around well 10. Another possibility is that the natural ground water could have actually contained concentrations of nitrogen forms at these levels; however, the data from well 4 tend to refute this possibility. Since well 4 was located on the north side of West Brook, it could not be contaminated by the sewage effluent; therefore, it was more representative of the natural ground water. Data from well 4 indicate the nitrate and ammonia concentrations were well below 0.10 mg/l, and the TKN concentrations did not exceed 1.0 mg/l.

As anticipated, the treatment plant had high TKN values ranging from 14.7 to 18.5 mg/l and 6.3 to 9.7 mg/l for the influent and effluent, respectively. The ammonia concentrations were also high but decreased during the treatment process. These decreases occurred as the ammonia and TKN were converted to nitrates and were most evident in the summer season when an increase in nitrates in the effluent occurred. Some of the reduction was a result of partial removal of organic forms in the treatment processes.

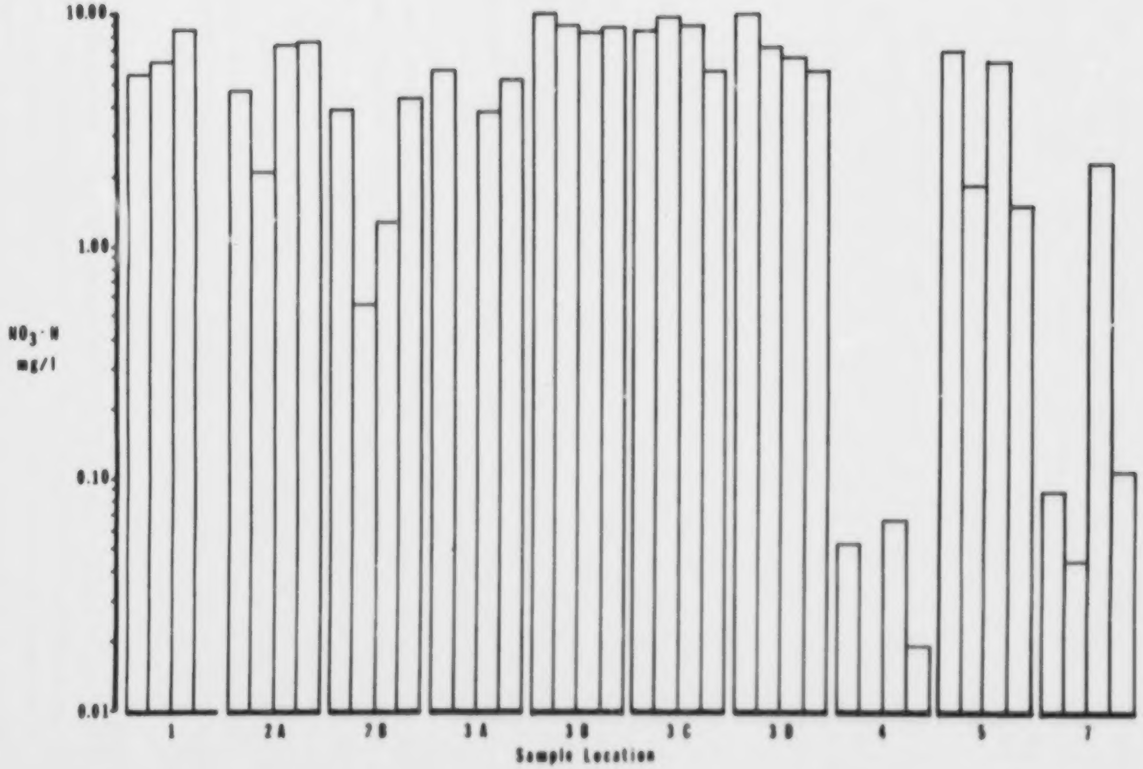


Figure 34A. Seasonal Variations in Nitrate. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

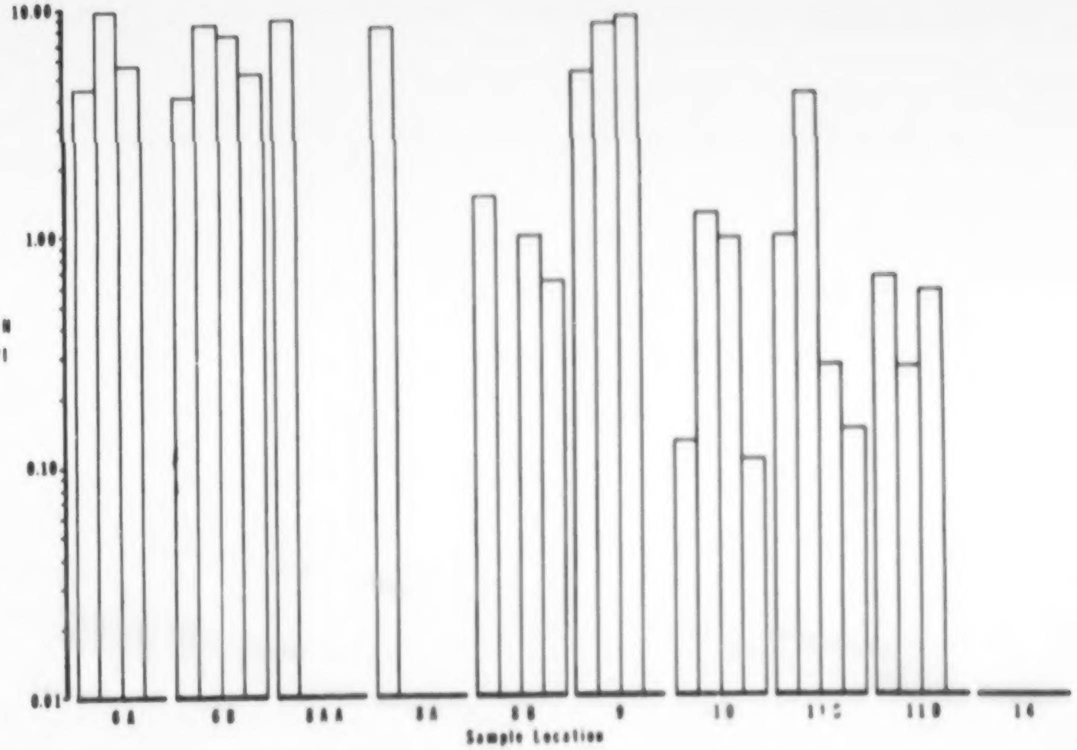


Figure 34B. Seasonal Variations in Nitrate. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

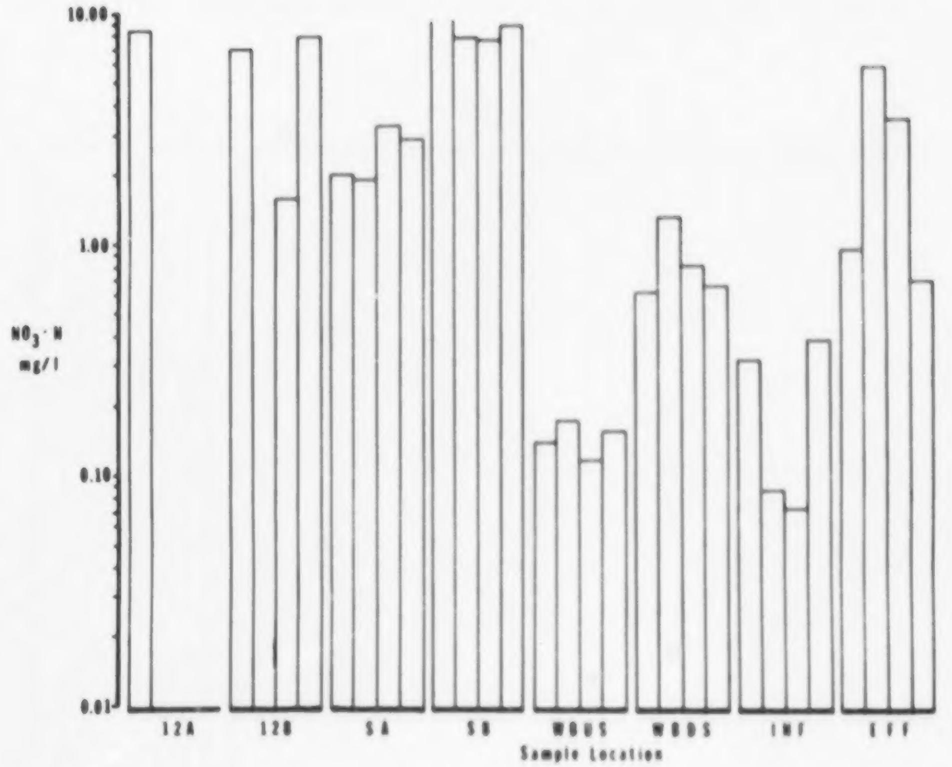


Figure 34C. Seasonal Variations in Nitrate. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

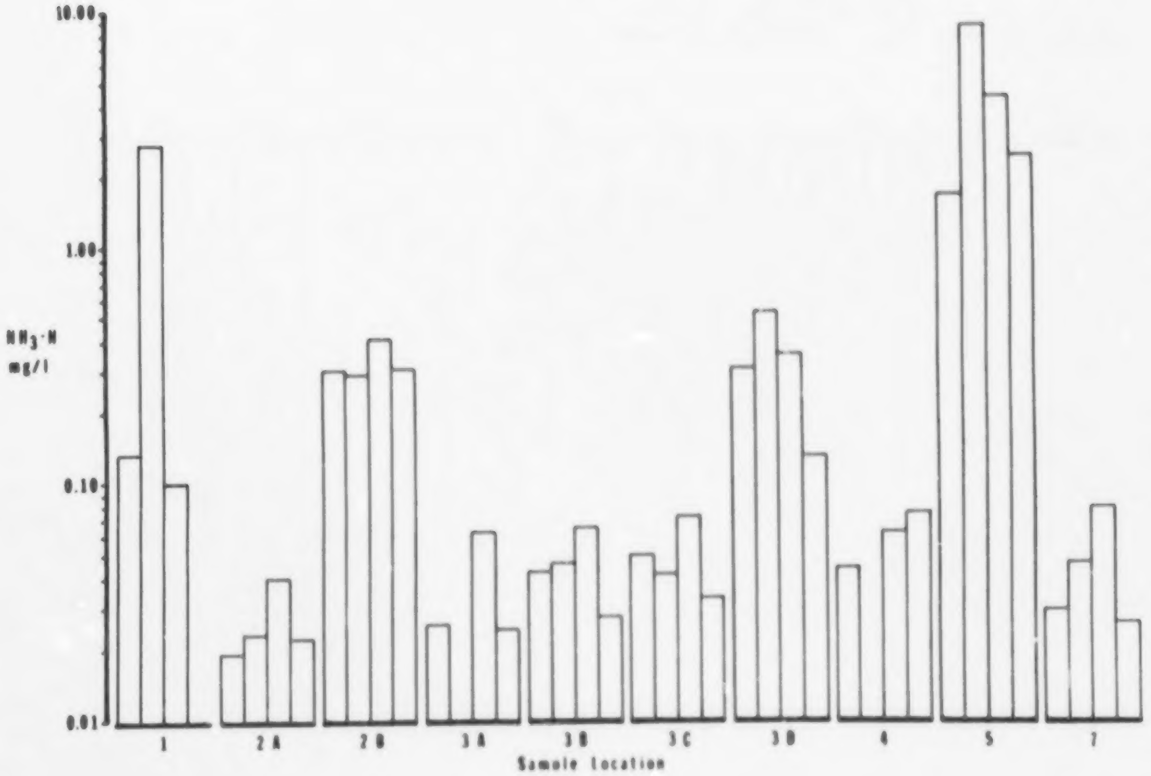


Figure 35A. Seasonal Variations in Ammonia Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

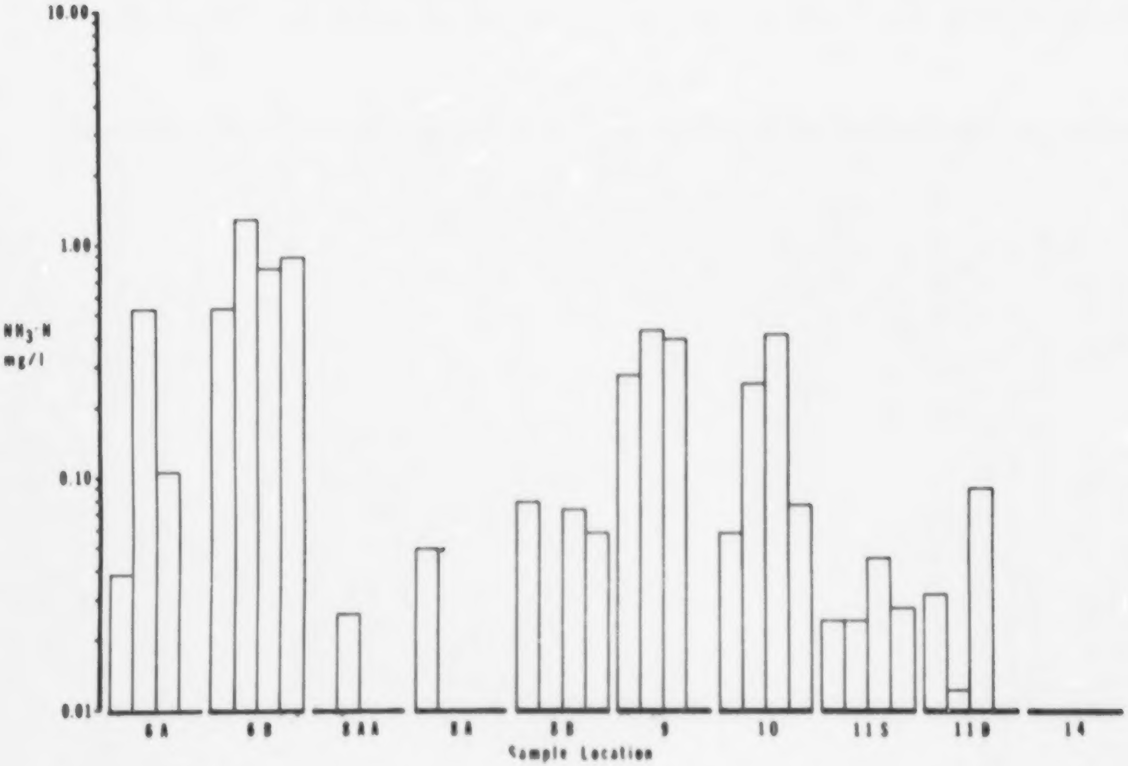


Figure 35B. Seasonal Variations in Ammonia Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

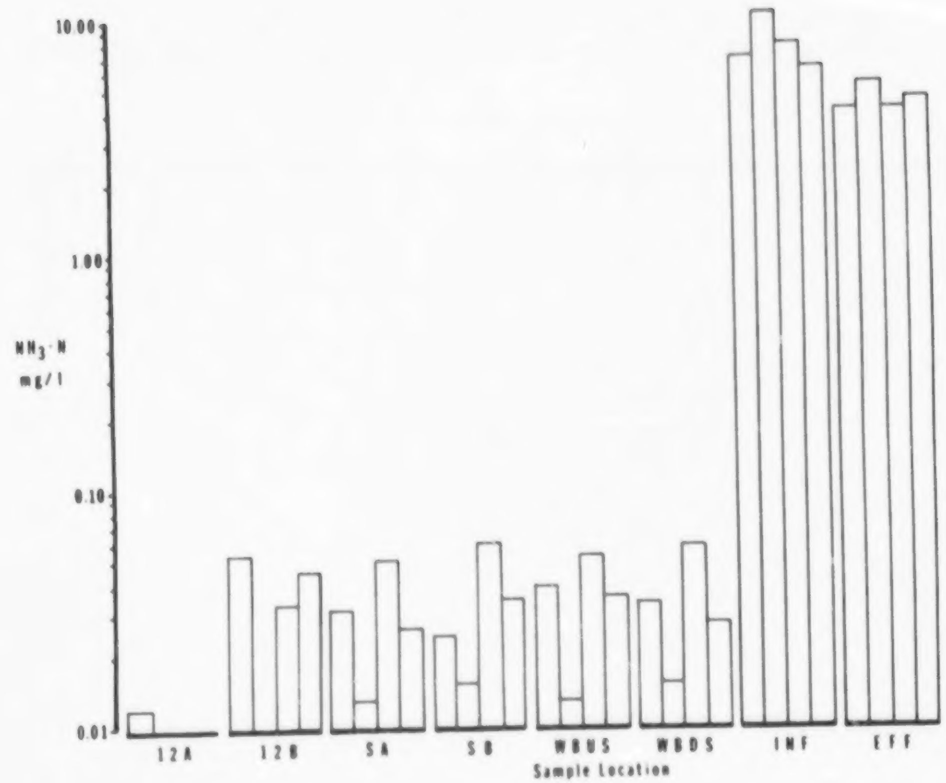


Figure 35C. Seasonal Variations in Ammonia Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

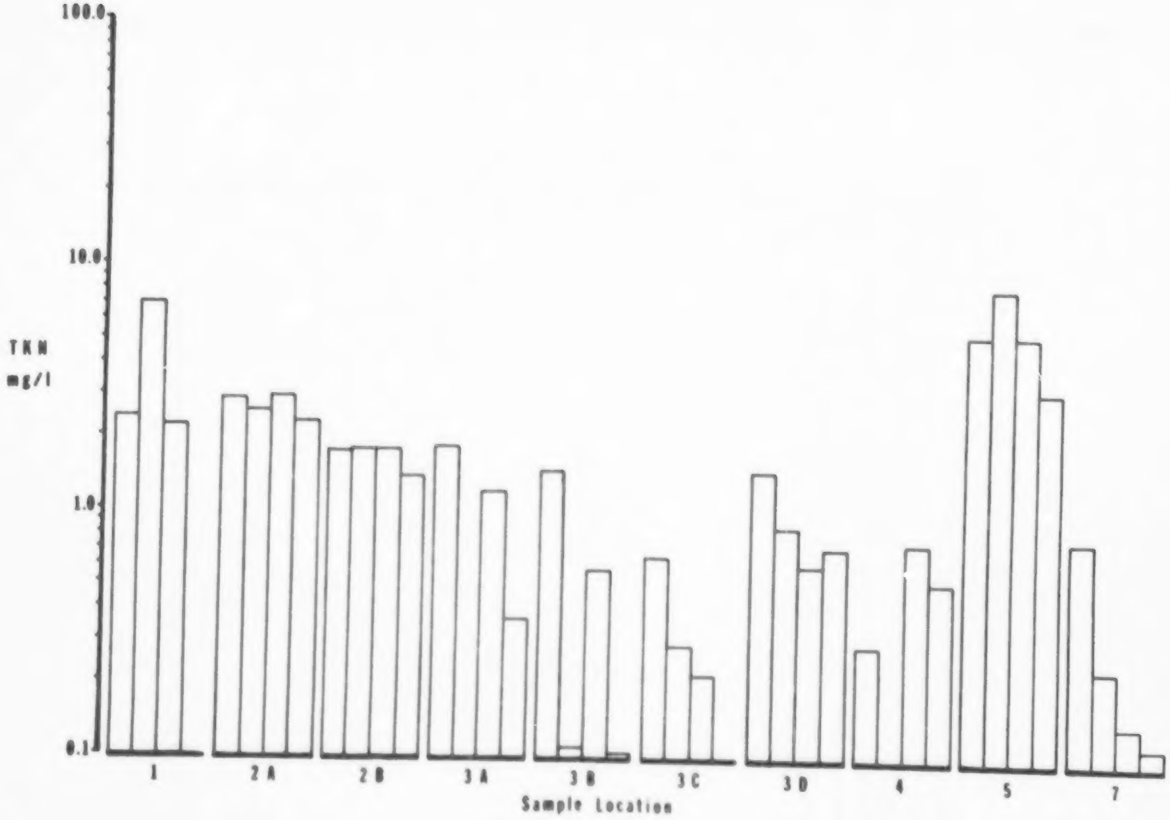


Figure 36A. Seasonal Variations in Total Kjeldhal Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

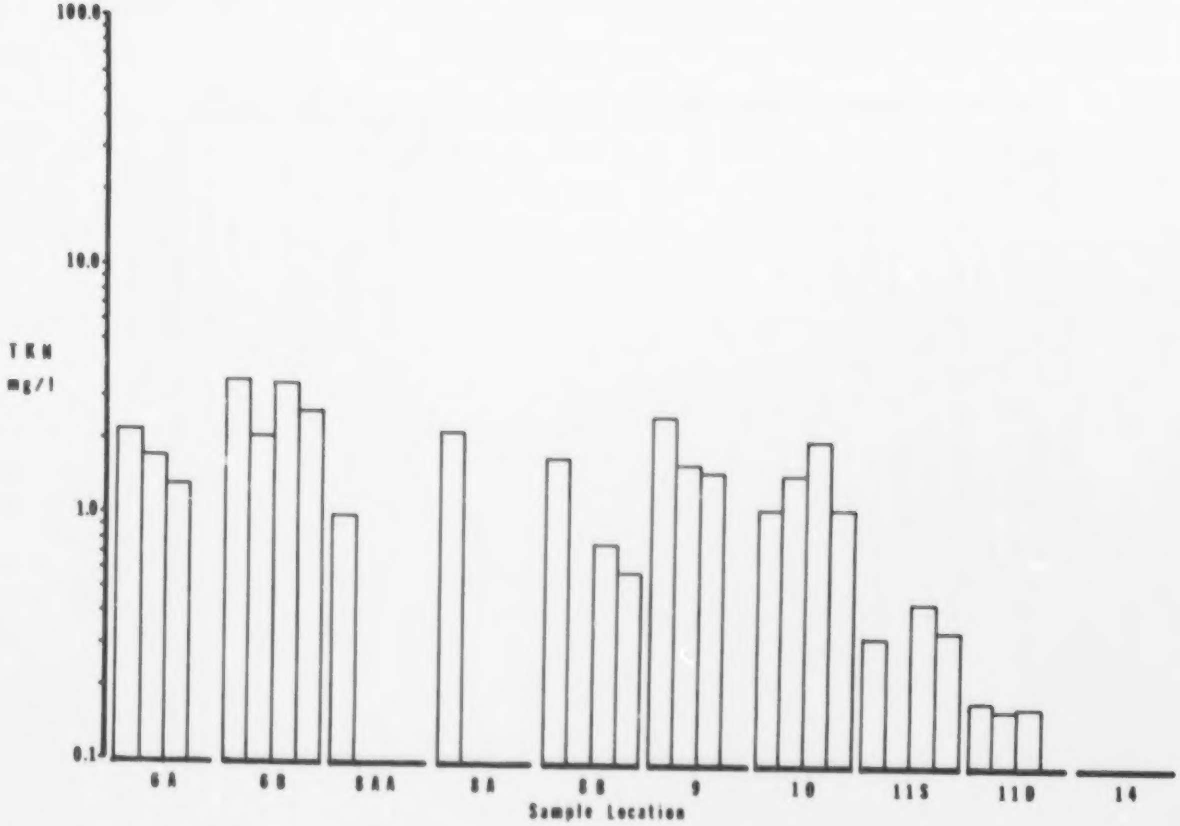


Figure 36B. Seasonal Variations in Total Kjeldhal Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

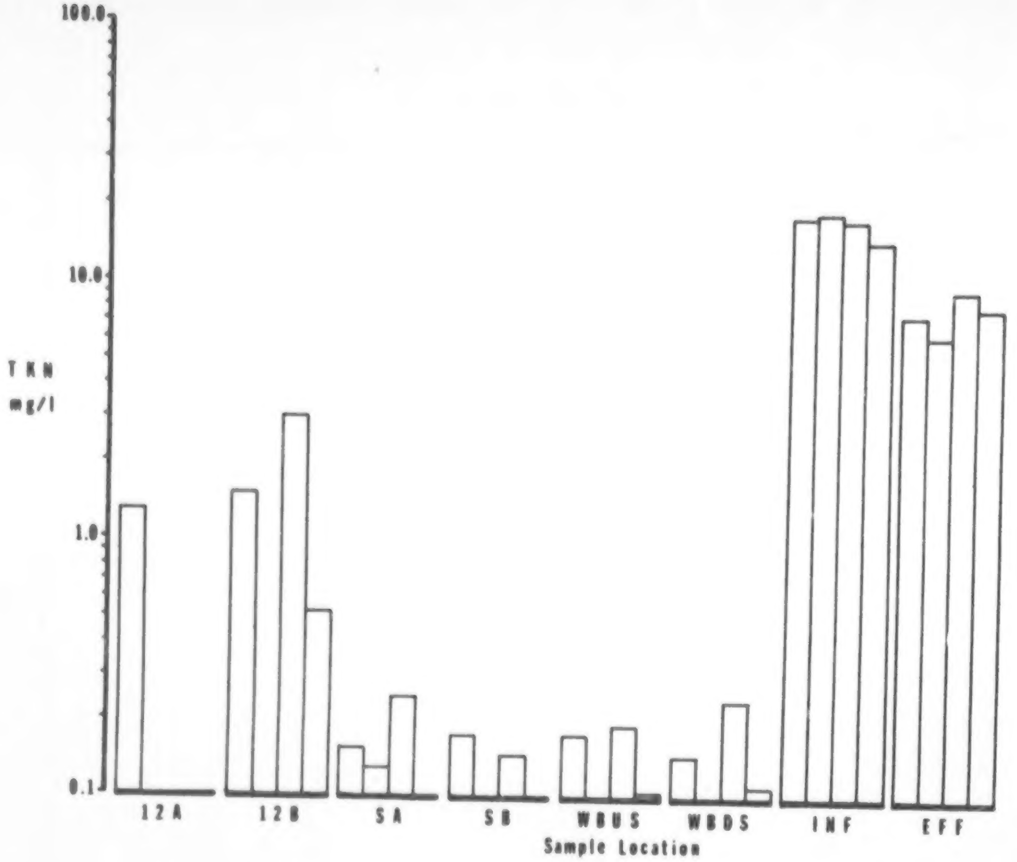


Figure 36C. Seasonal Variations in Total Kjeldhal Nitrogen. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

During the vertical transport of wastewater applied to the sand beds, the TKN and ammonia concentrations decreased only slightly but the nitrates generally increased, particularly in the spring, fall, and winter as shown in wells 1 and 5. However, wells 11S and 11D had nitrate concentrations generally below 1.0 mg/l, except for well 11S in the summer when it reached 4.0 mg/l. Wells 8AA and 9, downgradient from the treatment plant, exhibited nitrate levels of over 5.0 mg/l and TKN levels of 1.0 mg/l. The ammonia concentrations in well 9 were above 0.25 mg/l, but the concentrations in the well 8 series were not as strongly influenced by the sewage percolate as in well 9. These results indicate that the well 8 cluster is located on the fringe of the laterial sewage flow through the saturated zone. Nitrate observations further downgradient at the well 6 series were about 4.0 mg/l while the ammonia levels were higher than those at well 9 (6B was above 0.50 mg/l), and TKN values were roughly the same in wells 6 and 9. This suggests that denitrification of the percolating wastewater occurred during its laterial movement between wells 9 and 6. In comparison of wells further down gradient, the well 12 series experienced high nitrate and low ammonia concentrations with TKN values slightly lower than those in the well 6 series.

Wells 2 and 3 are the farthest from the sand beds and are located about 400 to 500 m (1,300-1,650 ft) downgradient. Well 2A had slightly higher nitrate levels than 2B, indicating the higher DO concentrations allowed oxidation of ammonia and organic forms. The TKN levels in 2B were only slightly lower than 2A, but the ammonia values in 2B were an order of magnitude higher than those in 2A, 0.04 mg/l, and .40 mg/l, respectively. These data may be correlated with the low DO concentrations found in well 2B. By the time the effluent reached well 3, the nitrate concentration was almost 10 mg/l, the upper limit of the drinking water standards (21). The nitrate concentrations in the upper portion of the aquifer, as shown by well 3A, remained below 6.0 mg/l which was probably due to the influence of runoff and precipitation. The ammonia levels in the lower portion of the aquifer as illustrated by data from well 3D reached 0.66 mg/l in the summer, confirming other findings which indicate that the sewage flow concentrates in the lower regions of the aquifer. The lower seepage streams had minimum nitrate levels of 7.50 and 7.70 mg/l in the fall and summer, respectively. The upper seepage stream had maximum nitrate levels of 3.27 and 2.84 mg/l in the fall and winter, respectively. These data seem to indicate that the seepage below Gage Road was much more strongly influenced by the effluent than the seepage above Gage Road. The impact of the seepage streams flowing into West Brook could be readily seen when comparing the upstream and downstream stations. The nitrate concentrations in West Brook increased from a range of 0.11 to 0.17 mg/l at the upstream station to a range of 0.60 to 1.30 mg/l at the downstream station.

General observations and interpretation of data have led to the conclusions that the nitrogen entering the treatment plant was primarily as organic and ammonia nitrogen which was partially converted to nitrate as the effluent passed through the treatment system, soil, and aquifer. However, well data indicated that denitrification occurred in the zones of the soil and aquifer. By the time the effluent emerged from the subsurface to form the seepage streams, the nitrogen had essentially been converted to the nitrate form.

The high nitrate concentration of the seepage streams was diluted approximately an order of magnitude in West Brook, resulting in a yearly average nitrate level of about 0.73 mg/l entering Lake George. Vollenweider recommended an upper limit of 0.3 mg/l to control excessive algal growths (24).

Due to the wide range of values encountered at the diverse sampling locations in this study, a log scale was used for bar graphs depicting seasonal changes in phosphorus compounds. Figures 37A, 37B, and 37C illustrate seasonal changes in soluble reaction phosphorus (SRP) at all stations while Figures 38A, 38B, and 38C illustrate seasonal changes in total phosphorus (TP). The treatment plant influent and effluent had high levels of both SRP and TP. Well data presented indicate the degree of phosphorus removal as wastewater percolates through the sand beds. Well 1 exhibited a slight drop in SRP levels from the effluent, but SRP levels in well 5 actually increased to 0.70 mg/l in winter and 3.80 mg/l in summer. Polyphosphates (TP) also showed an increase in well 5 but not so pronounced as SRP. The high values in well 5 could be the result of a release of absorbed or precipitated phosphates in the shallow unsaturated soil profile due to pH changes or redox potential changes. Wells 4, 7, and 10 indicated that the natural ground water had SRP concentrations within a range of 0.01 to 0.01 mg/l. The SRP levels at the well 6 series (particularly at well 6B) dropped to less than 0.02 mg/l. The SRP levels increased to 0.05 mg/l at the well 12 series and then decreased to less than 0.01 mg/l at the well 2 series. At the well 3 series, the two shallowest points had SRP concentrations comparable to those in control well 4. Well 3C, however, had much higher levels of SRP than the shallower elevations, and well 3D at the bottom of the aquifer had very low levels. The seepage streams had SRP concentrations approximately the same as native ground water, causing no significant change in the SRP concentration of West Brook.

In general, those well locations which exhibited high values for ammonia concentrations had very low SRP concentrations. The two conditions are apparently related, but conditions which favor the formation of ammonia (low DO and low redox potential) should inhibit precipitation of phosphates. Instead, it appears that the orthophosphates were efficiently removed under those conditions. Total phosphorus concentrations were relatively high in wells 1 and 5 indicating that there was little TP removal in sand beds N-4 or S-3. Well 9 showed no decrease in TP concentrations, but the well 8 series exhibited depressed TP levels along with the well 6 and 12 series. Although TP concentrations at well 2B were approximately the same as those of native ground water, concentrations in the upper portion of the saturated aquifer at well 2A ranged from 1.80 mg/l in the spring to 2.50 mg/l in the winter. Wells 3A and 3B, also located in the upper portion of the aquifer, had TP levels slightly above those of native ground water, but the deeper wells, 3C and 3D, had TP concentrations about the same as control well 4. Comparisons of TP measurements between the West Brook upstream and West Brook downstream indicated there was an insignificant contribution of TP to West Brook from the infiltration system. Examination of data from the control wells indicated well 7 appeared to be definitely contaminated from surface dumping of sand bed scrapings while well 10 had TP concentrations only slightly above those in control well 4.

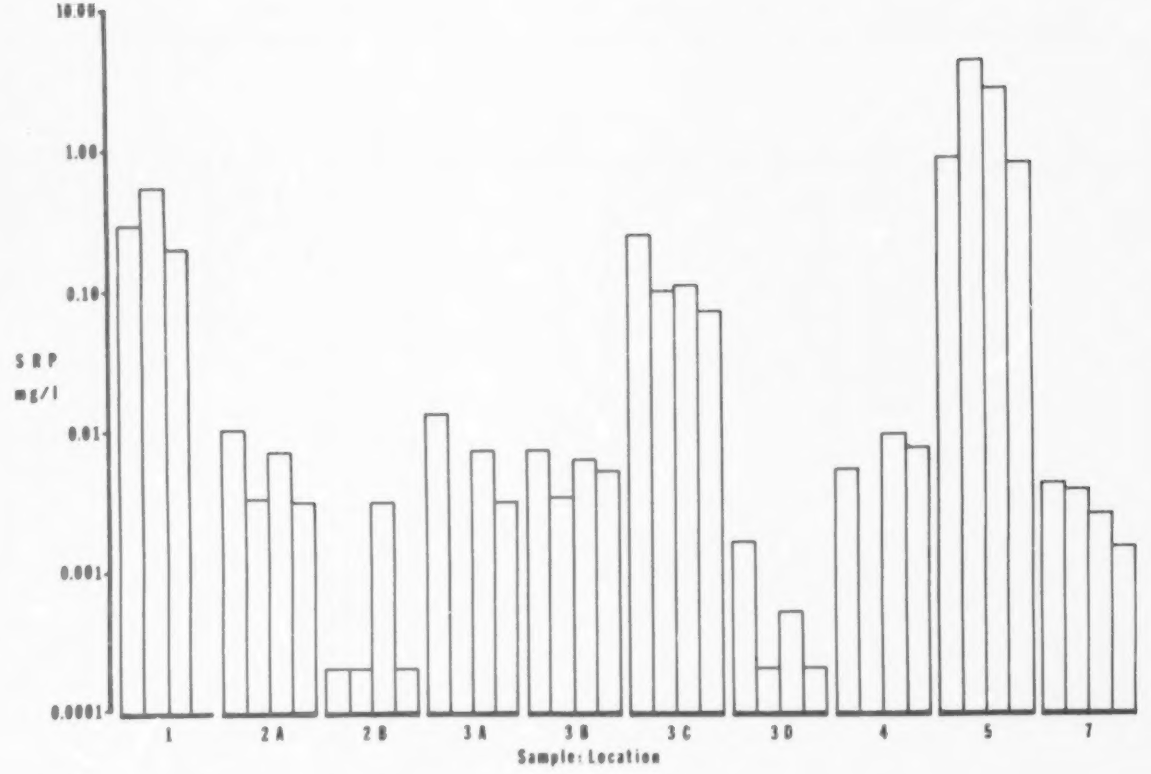


Figure 37A. Seasonal Variations in Soluble Reactive Phosphate-Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

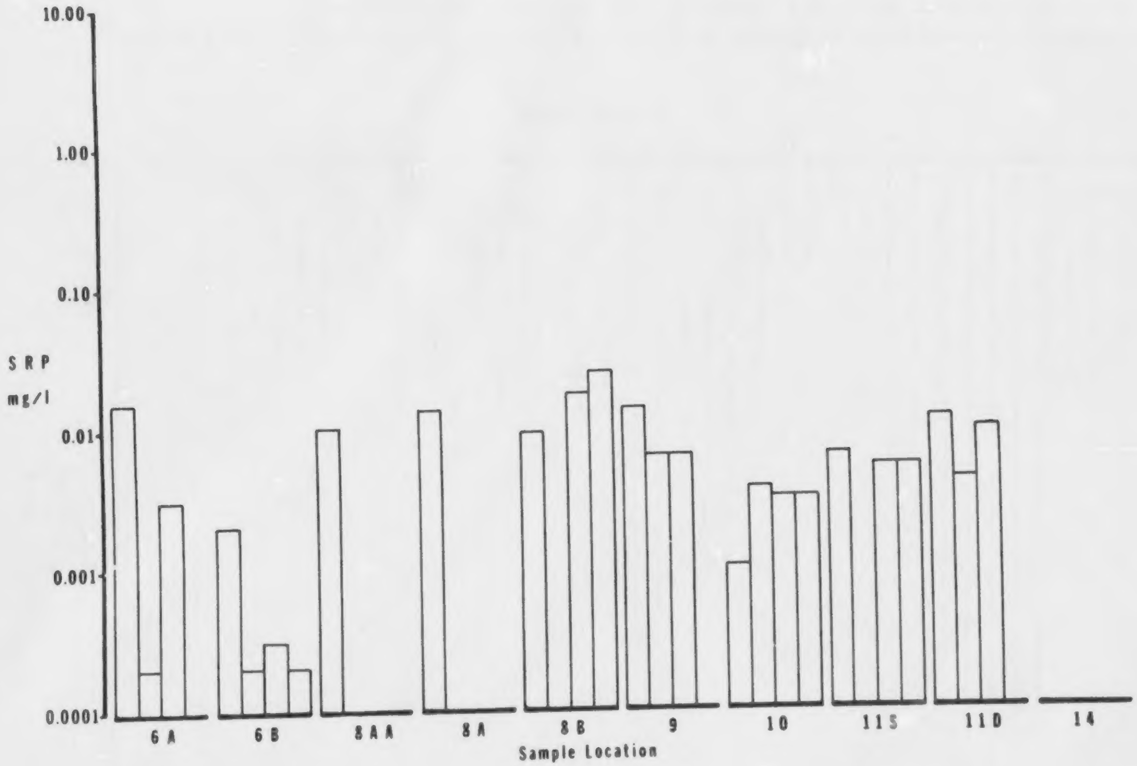


Figure 37B. Seasonal Variations in Soluble Reactive Phosphate-Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

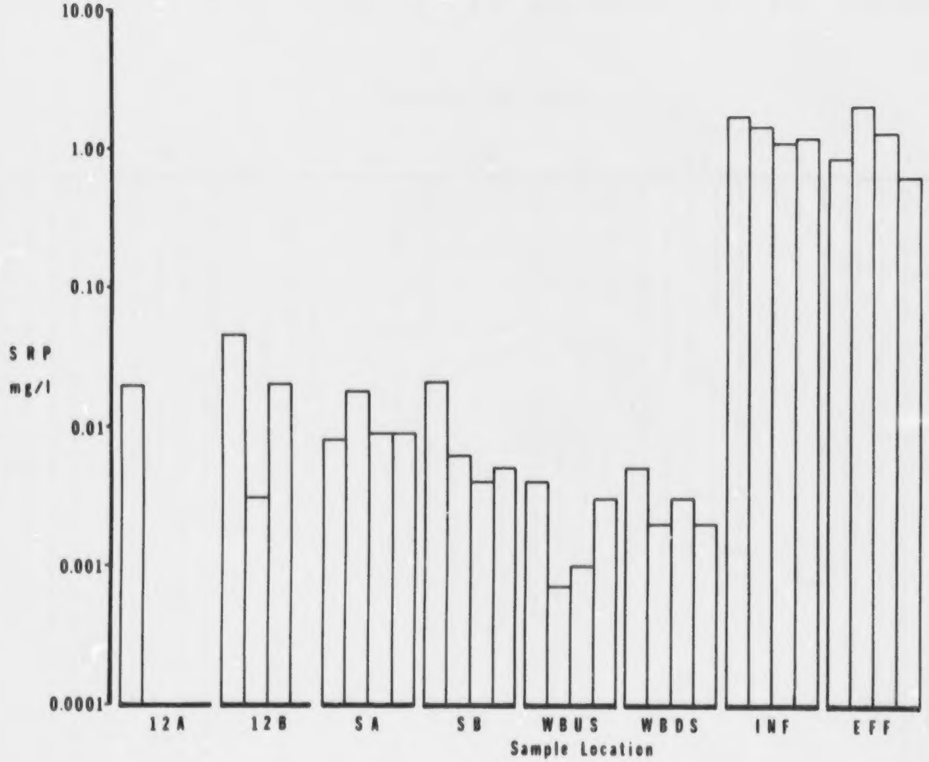


Figure 37C. Seasonal Variations in Soluble Reactive Phosphate-Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

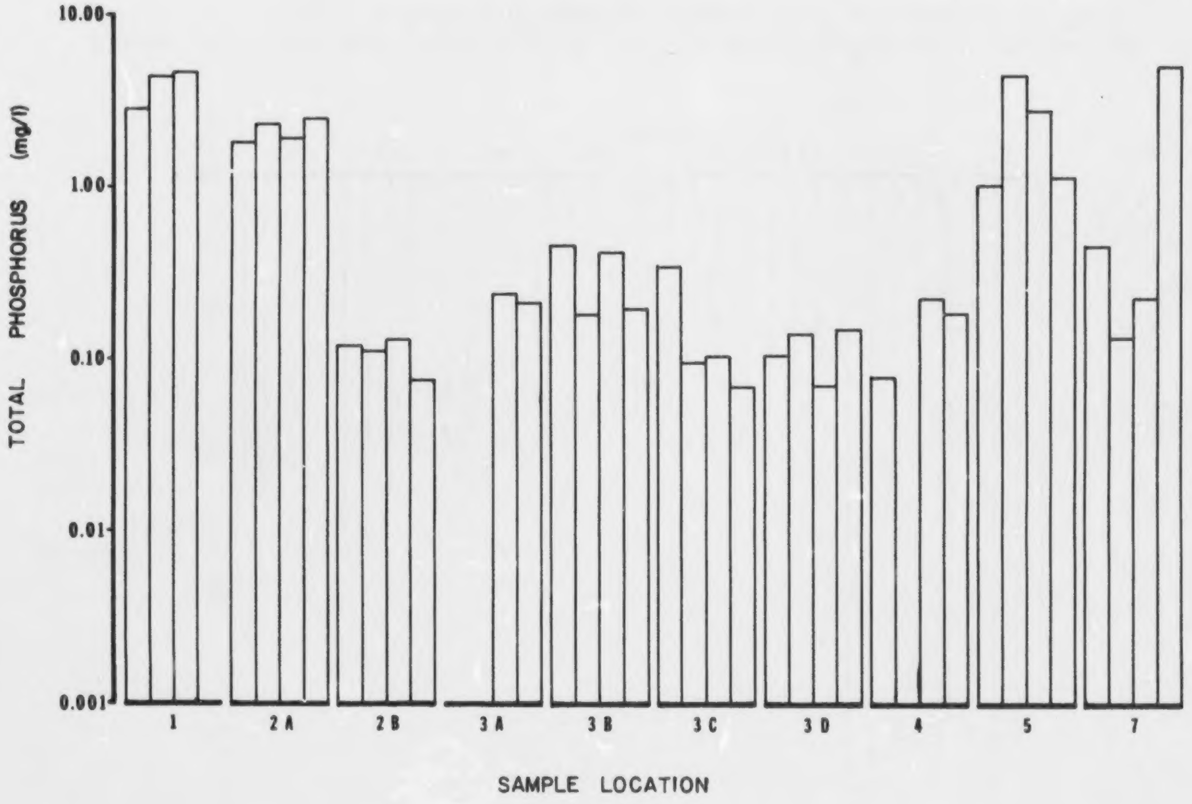


Figure 38A. Seasonal Variations in Total Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

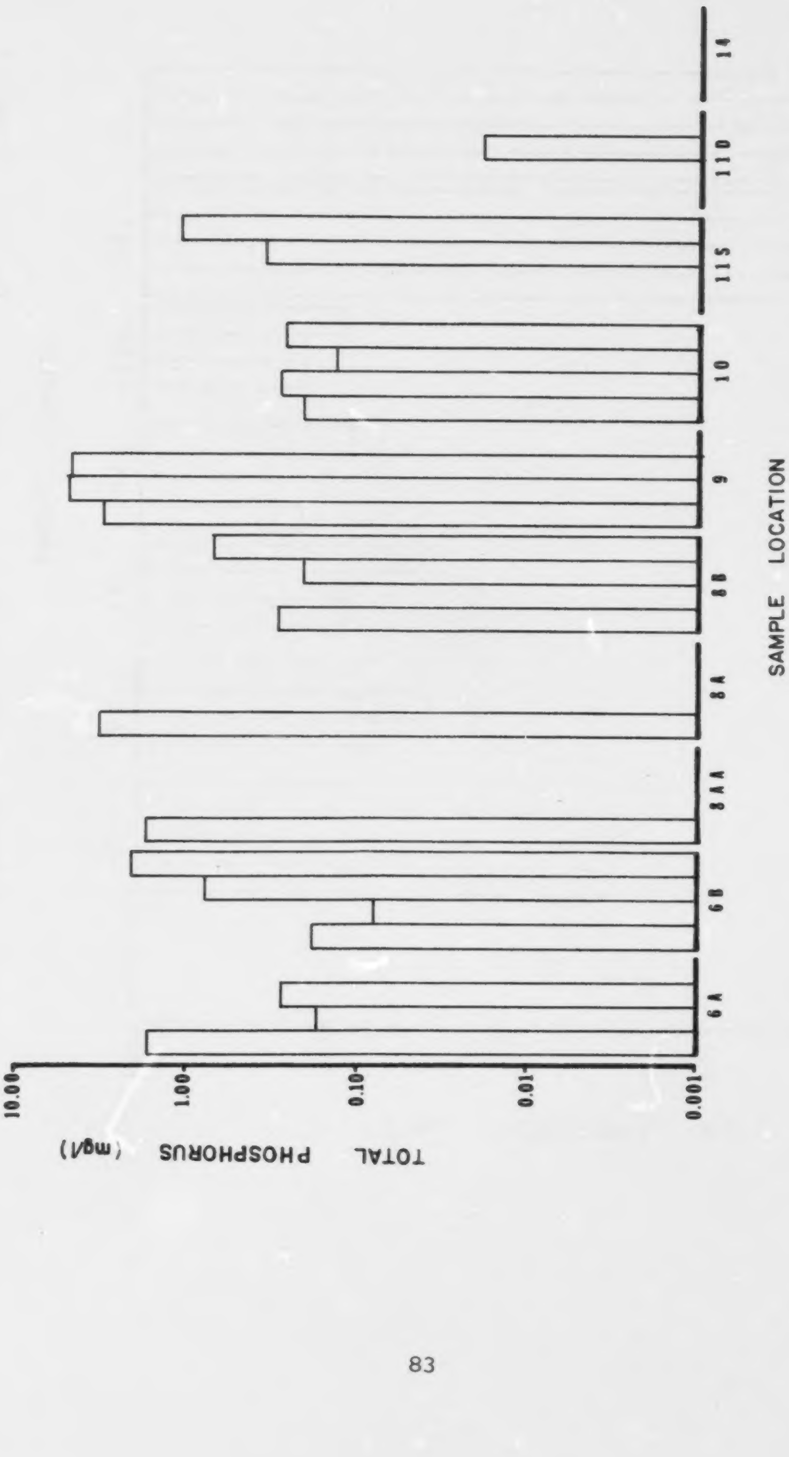


Figure 38B. Seasonal Variations in Total Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

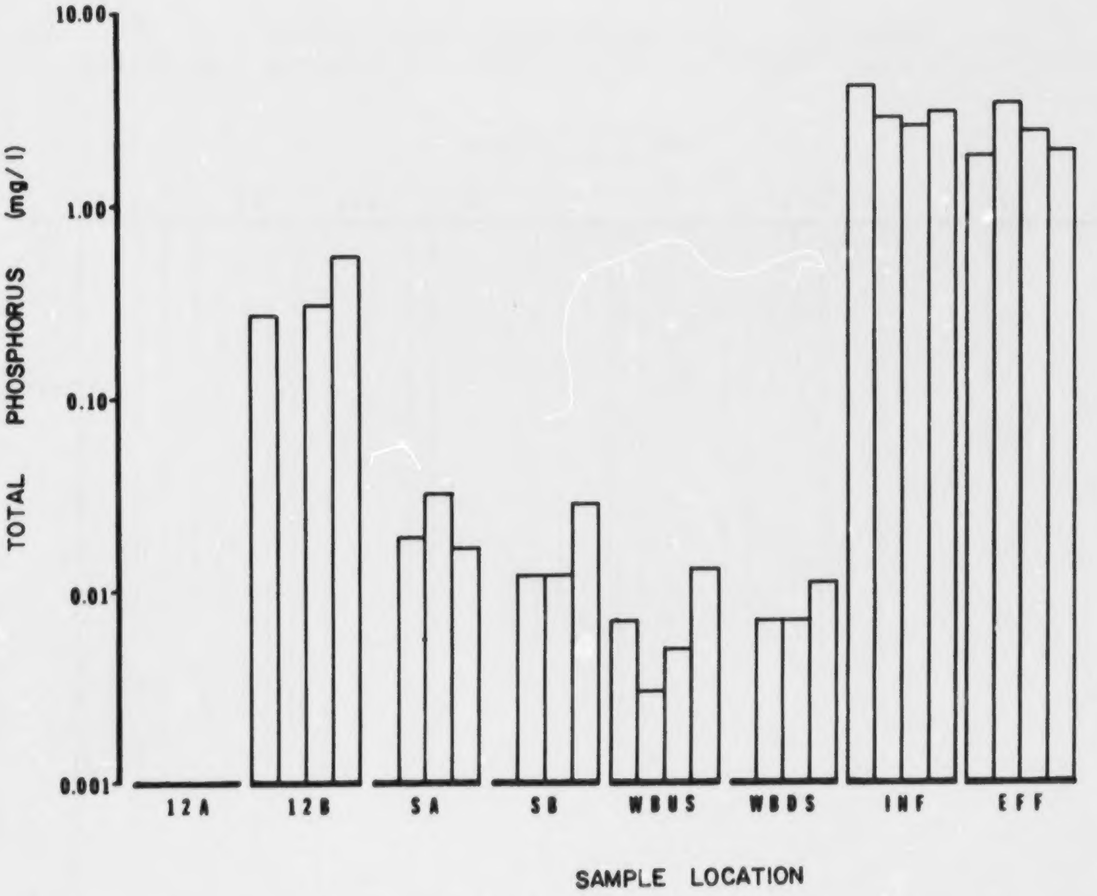


Figure 38C. Seasonal Variations in Total Phosphorus. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

Seasonal averages of alkalinity are presented in Figures 39A, 39B, and 39C. The native ground water had an alkalinity ranging from 40 mg/l in well 10 to 60 mg/l in well 4. Data indicate the alkalinity of the effluent was somewhat seasonally dependent, but to a much lesser degree than the influent. Both had alkalinities ranging from about 80 to 170 mg/l. Well 1 had an alkalinity of about 100 mg/l while well 5 had values as low as 61 mg/l in the fall. Wells 5 and 7 experienced very similar seasonal fluctuations. Well 8A had an unexplained high alkalinity in the summer while wells 8AA, 9B, and 9 had values around 100 mg/l. The well 11 series appeared to have alkalinities about the same concentration as control well 4. Proceeding downgradient from well 6, a general increase in alkalinities was evident with the highest values occurring in well 3 and station SB. The high alkalinity from the seepages increased the values in West Brook from about 25 mg/l to about 40 mg/l.

Chlorides (Cl) are known to be present in sewage and are a good indication of water course contamination. The seasonal averages of chloride concentrations are plotted in Figures 40A, 40B, and 40C according to the previous conventional listing of sampling sites. The treated sewage chloride concentration fluctuated from about 65 to 101 mg/l, with the highest concentration occurring in the winter and the lowest in the fall. The concentration in wells 1 and 5 was approximately one-half that of sewage applied to the sand beds. High chloride levels were observed in wells 2A, 2B, 3B, 3D, 6A, and 6B. The high chloride concentrations in the deeper wells at site 3 were attributed to infiltration of the stored road deicing salt at the town garage, while wells 2 and 6 high concentrations are a result of the infiltrating sewage moving through the aquifer. Moderate chloride levels were observed in wells 4, 8AA, 8A, 8B, 9, 11S, and 11D which can also be attributed to the infiltration of sewage in the sand beds. Low values were found in wells 7, 12A, and 12B. The lowest values were in control well 10 which never exceeded 5 mg/l. The seepage above Gage Road had a moderate chloride concentration indicating dilution by native ground water while the well 3 series and station SB had a high chloride level indicating the road deicing salt contamination had not been diluted by the time it had reached these stations. High chloride concentration from the seepage areas did reflect an increase in chloride in West Brook as can be seen by comparing data from stations WBUS and WBDS.

The dissolved solids concentrations of water samples from all the stations were monitored regularly and used to indicate the degree of sewage contamination. Figures 41A, 41B, and 41C show the average seasonal dissolved solids concentrations at all the sampling locations. There seemed to be very little seasonal fluctuation in the concentration of dissolved solids; however, the location of the sewage plume can be fairly well defined by study of data from all the stations. The highest values were observed in the below Gage Road station and in the multiple depth wells at the well 3 site. These high values can be attributed to the influence of the aforementioned highway deicing salt. An unexplained high value was also noted in well 6A during the summer. Average values of about 200 mg/l were found in the treatment plant influent and effluent, in wells 1, 2A, 2B, 5, 6B, 8AA, 9, 12A, and 12B, and in the seepage above Gage Road. Lower values were found in wells 4, 7, 8B, 10, 11S, and 11D. There was a slight increase in the dissolved solids in West Brook between the upstream and downstream stations.

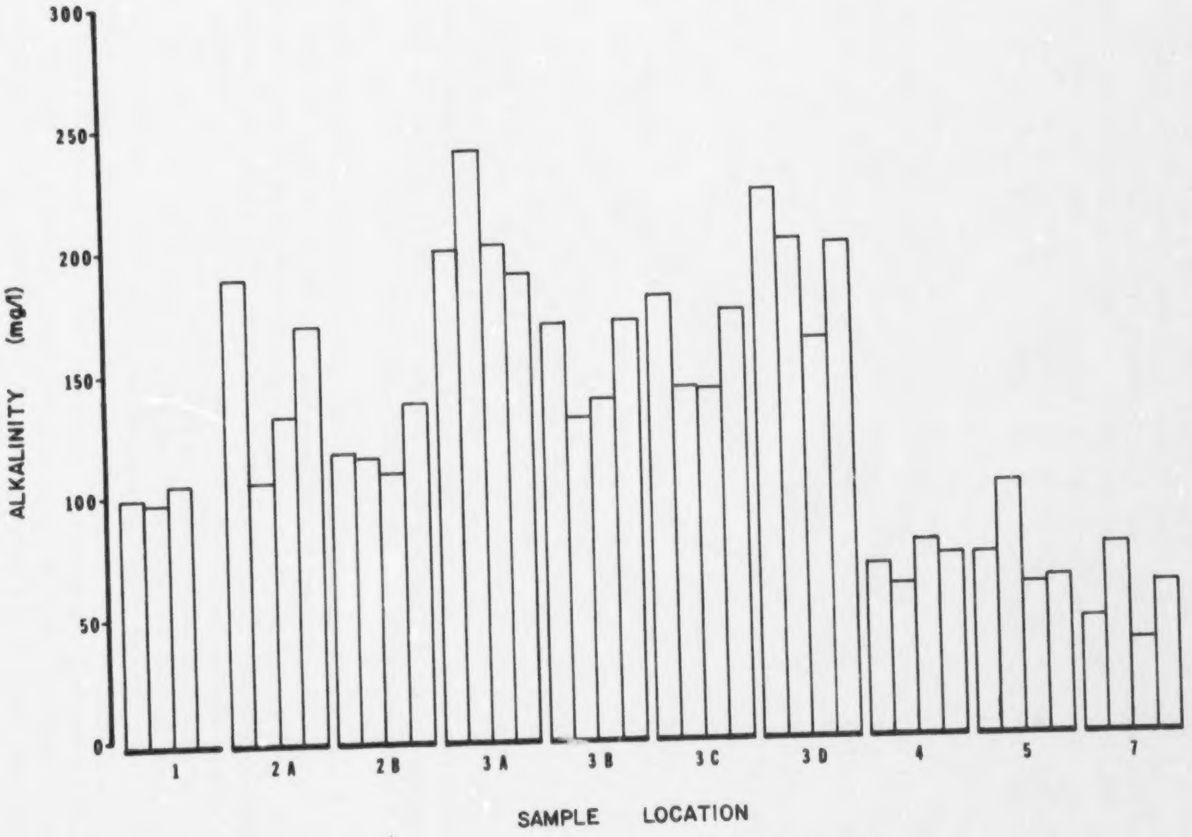


Figure 39A. Seasonal Variations in Alkalinity. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

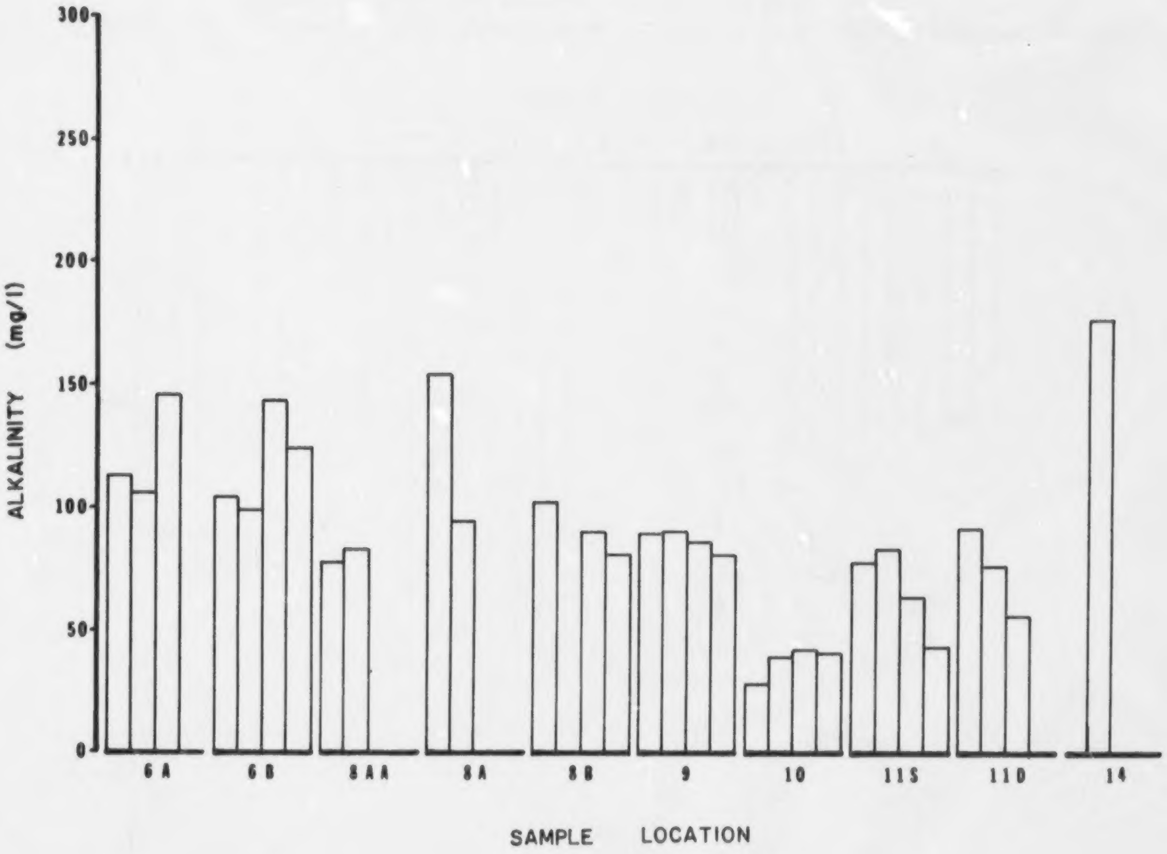


Figure 39B. Seasonal Variations in Alkalinity. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

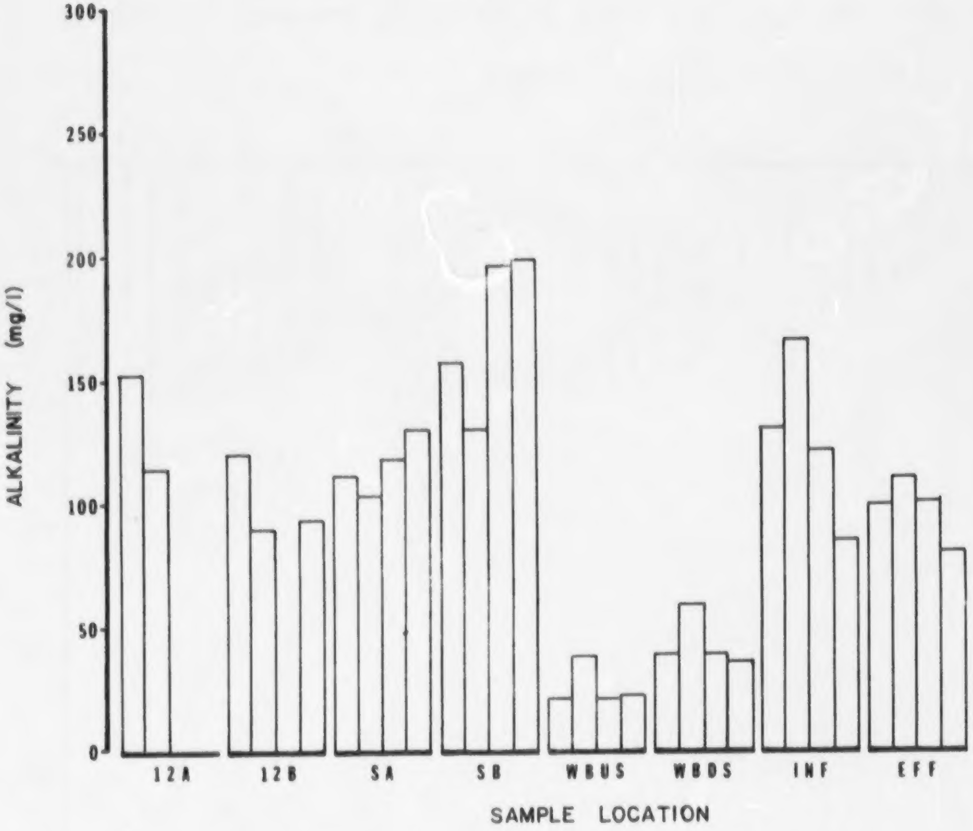


Figure 39C. Seasonal Variations in Alkalinity. The Bars Represent Spring, Summer, Fall and Winter, in that Order.

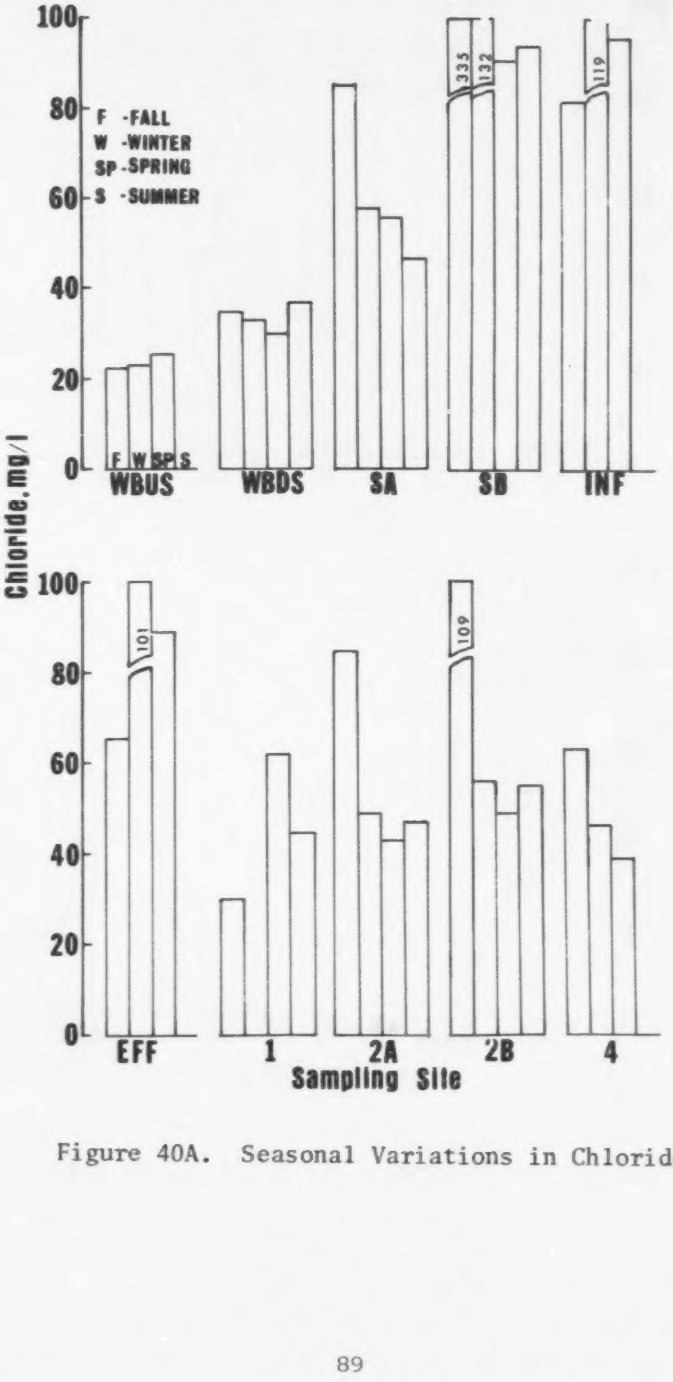


Figure 40A. Seasonal Variations in Chloride.

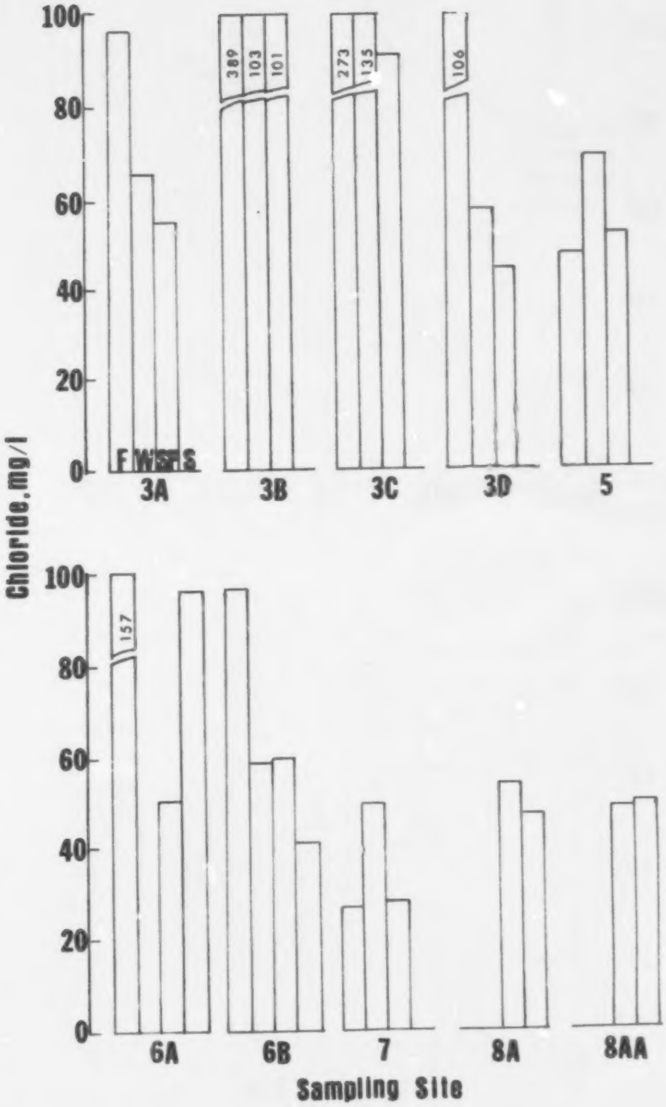


Figure 40B. Seasonal Variations in Chloride.

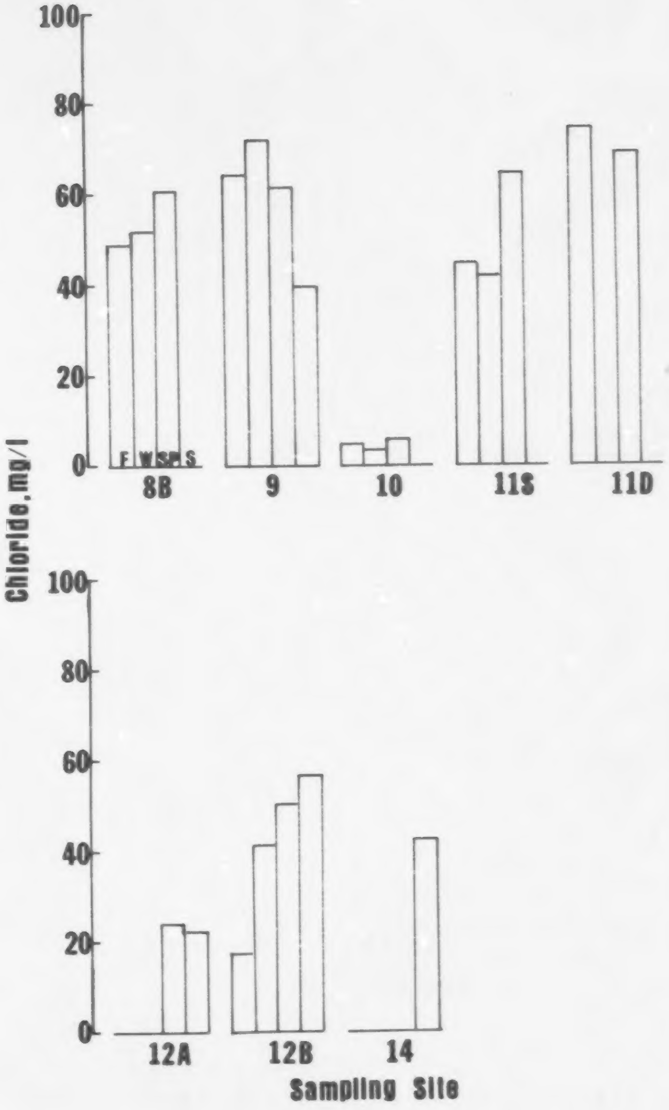


Figure 40C. Seasonal Variations in Chloride.

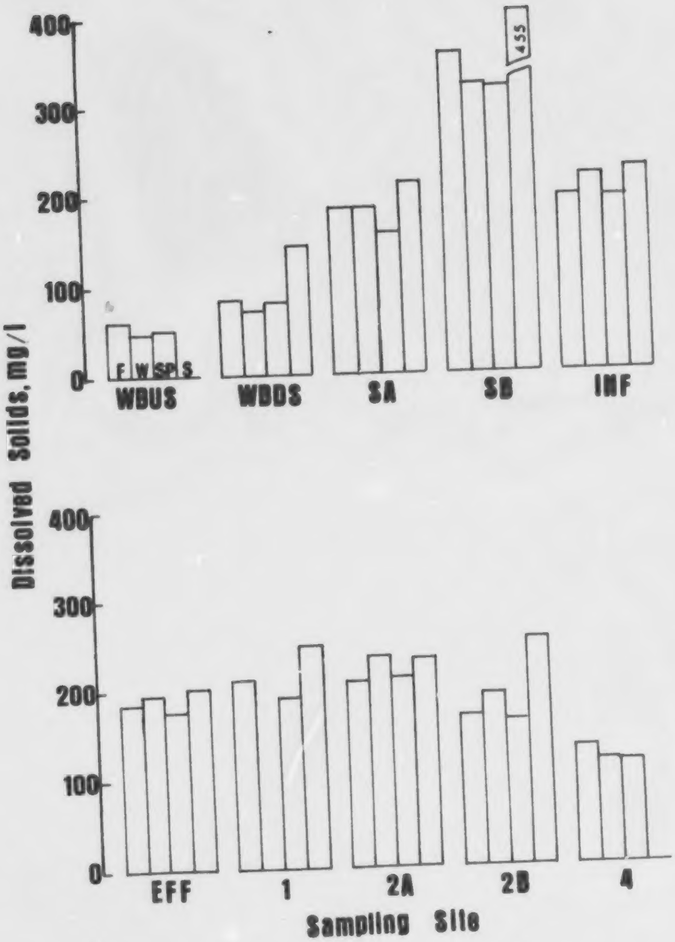


Figure 41A. Seasonal Variations in Dissolved Solids.

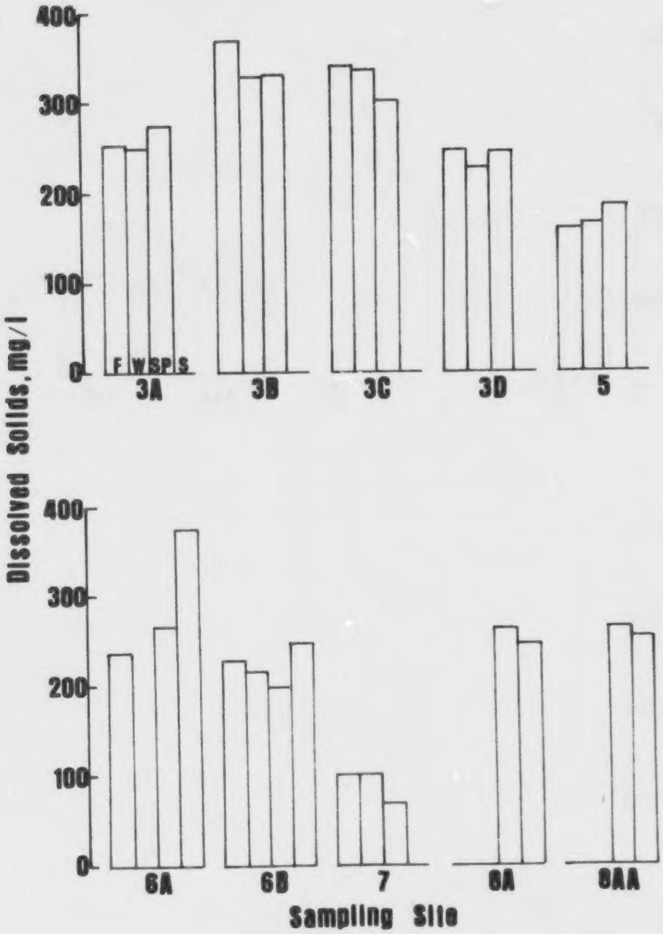


Figure 41B. Seasonal Variations in Dissolved Solids.

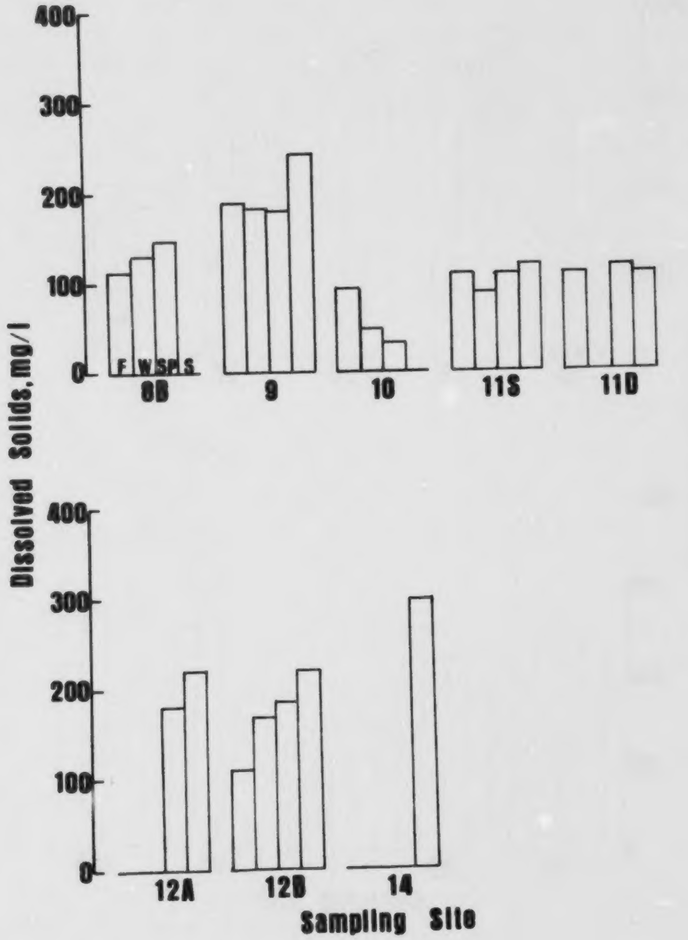


Figure 41C. Seasonal Variations in Dissolved Solids.

The biological parameters studied were not collected through all seasons, thus the data are presented in tables instead of seasonal averages on bar graphs. Table 2 contains daily results of fecal coliform analysis and average values for the station. As can be seen in Table 2, a 33 percent reduction in fecal coliform concentrations was achieved through the primary and secondary treatment process. The only wells which indicated definite signs of contamination were wells 1 and 5, both of which were located in sand beds. Well 5 was only about 4 m (13 ft) below the the surface of the sand bed and well 1 was almost 20 m (65 ft) below the sand bed surface. The coliforms should have been filtered and adsorbed before they reached either of these well points. Possible explanations are that the bentonite seal around the well tubing was ruptured allowing the effluent to seep down the side of the tubing. Another possibility is that the tubing may have become cracked during installation and effluent may have seeped into the well through these cracks. All other wells were essentially free of contamination. Those few organisms which did appear were probably the result of bailer contamination during sampling rather than actual presence of coliforms in the wells.

The surface waters contained significant numbers of fecal coliforms which are apparently a result of contamination from animals. A number of reconnaissance tours were made of West Brook. These tours extended upstream for several miles through the forest and the only sources of contamination which were found were some trails frequented by horses from a nearby stable. These trails were located approximately one mile upstream of the seepage sites. A small abandoned reservoir once used to store water for drinking supplies for the Village was also observed in the same area. The number of coliform organisms present in the surface waters seemed to be dependent on the amount of runoff occurring at the time the samples were collected. For example, samples collected on August 10 after a heavy rainfall in the area were high in coliforms. Heavy rains also occurred on February 2 and 10 and on June 16 and 20. Samples collected within two days of these rainfall events also had high coliform counts. Therefore, the data suggest that non-point sources were responsible for at least a portion of the coliform organisms present in West Brook.

Total coliforms showed the same basic trends as fecal coliforms. The total coliform data presented in Table 3 also indicate contamination of wells 1 and 5. Surface waters contained low numbers (generally about 50 organisms/100 ml) of coliforms. The treatment plant was effective in reducing total coliforms in the influent by about 25 percent as shown by the effluent applied to the sand beds.

The presence of fecal streptococci species is an indication of fecal waste contamination by warm-blooded animals. Table 4 contains the data for the enterococci determinations. Enterococci are streptococci normally found in the intestinal tract of warm blooded animals. In general, results were very similar to coliform measurements; however, well 5 indicated no evidence of enterococci contamination. The surface waters exhibited low but measureable contamination similar to coliform studies. This contamination was probably also from non-point sources.

The virus studies were limited to measurement of only coliphage. These viruses were collected at all stations on three separate occasions and summarized as shown in Table 5. All wells, including the control wells, indicated the presence

TABLE 2. FECAL COLIFORMS (100 ml^{-1})

Date	1	2A	2B	3A	3B	3C	3D	4	5	6A	6B	7	8AA	8A
1/6	-	-	-	-	0	0	0	-	-	-	0	-	-	-
1/16	-	0	0	-	0	0	0	-	-	-	-	-	-	-
1/28	-	-	-	-	-	-	-	-	212	-	1	-	-	-
2/4	-	-	-	1	0	1	0	-	-	-	-	-	-	-
2/13	-	0	0	-	-	-	-	-	-	-	-	-	-	-
2/20	-	0	0	0	0	0	0	0	-	-	0	0	-	-
2/25	-	-	-	-	-	-	-	-	-	-	0	0	-	-
3/5	-	0	0	-	0	-	-	-	-	-	-	-	-	-
3/10	-	-	-	0	0	0	0	-	0	-	-	0	-	-
3/31	145	0	0	-	-	-	-	1	-	-	-	-	-	-
4/6	226	-	-	-	-	-	-	-	1	-	0	0	-	-
4/15	-	0	0	0	0	0	0	0	-	-	-	-	-	-
4/22	-	-	-	-	-	-	-	-	-	-	-	0	-	0
4/29	214	-	-	-	-	-	-	0	-	0	0	-	-	-
5/20	0	-	-	-	-	-	-	-	74	0	0	0	-	0
5/27	-	-	0	0	0	0	0	0	-	-	-	-	-	-
6/16	-	0	0	-	0	0	0	-	-	-	0	-	0	0
6/24	14	0	0	-	-	-	-	-	-	1	0	-	-	-
7/17	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8/10	4	0	0	-	-	-	-	-	-	0	0	-	-	-
8/18	-	0	0	0	0	0	-	-	-	0	0	0	-	-
Average	101	0	0	0.2	0	0.1	0	0.2	72	0.2	0.1	0	0	0

(continued)

TABLE 2. (continued)

Date	8B	9	10	11S	11D	12A	12B	14	SA	SB	WBUS	WBDS	INF	EFF
1/6	0	0	-	-	-	-	0	-	0	2	-	-	-	-
1/16	-	-	-	0	-	-	0	-	0	-	-	-	-	-
1/28	0	2	-	-	-	-	-	-	-	-	-	-	TNTC	TNTC
2/4	-	-	-	0	-	-	-	-	0	7	25	20	-	-
2/13	0	-	0	0	-	-	0	-	0	-	-	-	TNTC	TNTC
2/20	-	-	-	-	-	-	-	-	1	1	47	43	-	-
2/25	0	-	0	0	-	-	-	-	-	-	-	-	510*	100*
3/5	-	-	-	-	-	-	0	-	1	3	0	0	-	-
3/10	-	-	-	-	-	-	-	-	-	-	-	-	470*	350*
3/31	-	0	0	-	-	-	2	-	-	-	-	-	-	-
4/6	0	0	-	0	0	-	-	-	-	-	-	-	-	-
4/15	-	-	-	-	-	-	0	-	0	2	0	0	-	-
4/22	-	-	0	0	0	-	-	-	-	-	-	-	500*	220*
4/29	-	0	-	0	0	-	-	-	2	1	2	1	-	-
5/20	-	-	0	-	-	-	-	-	-	-	-	-	-	600*
5/27	-	-	-	-	0	0	0	-	-	-	-	-	-	-
6/16	-	0	-	0	0	-	0	-	4	62	28	29	300*	190*
6/24	-	0	-	-	-	0	0	0	9	51	58	69	-	-
7/17	-	-	-	0	0	-	-	-	-	-	-	-	640*	620*
8/10	-	0	-	0	0	0	0	0	4	163	12	23	-	-
8/18	-	0	-	0	0	-	-	-	0	1	0	0	780*	430*
Average	0	0.2	0	0	0	0	0.2	0	1.8	29	19	21	533*	359*

* Expressed as 1000's per 100 ml.

TABLE 3. TOTAL COLIFORMS (100 ml^{-1})

Date	1	2A	2B	3A	3B	3C	3D	4	5	6A	6B	7	8AA	8A
1/6	-	-	-	0	0	0	0	-	-	-	0	-	-	-
1/16	-	0	0	-	0	0	0	-	-	-	-	-	-	-
1/28	-	-	-	-	-	-	-	-	385	-	0	-	-	-
2/4	-	-	-	0	0	1	0	-	-	-	-	-	-	-
2/13	-	0	0	-	-	-	-	-	-	-	-	-	-	-
2/20	-	0	1	0	0	0	0	0	-	-	0	0	-	-
2/25	-	-	-	-	-	-	-	-	-	-	0	0	-	-
3/5	-	0	0	-	0	-	-	-	-	-	-	-	-	-
3/10	-	-	-	0	0	0	0	-	0	-	-	0	-	-
3/31	250	0	0	-	-	-	-	0	-	-	-	-	-	-
4/6	TNTC	-	-	-	-	-	-	-	2	-	0	0	-	-
4/15	-	0	0	0	0	0	0	0	-	-	-	-	-	-
4/22	-	-	-	-	-	-	-	-	-	-	-	0	-	0
4/29	200	-	-	-	-	-	-	0	-	0	0	-	-	-
5/20	0	-	-	-	-	-	-	-	0	0	0	0	-	0
5/27	-	0	0	0	0	0	0	0	-	-	-	-	-	-
6/16	-	0	0	-	0	0	0	-	-	-	0	-	0	0
6/24	0	0	0	-	-	-	-	-	-	0	0	-	-	0
7/17	-	-	-	-	-	-	-	-	-	-	-	-	-	-
8/10	0	0	0	-	-	-	-	-	-	0	0	-	-	0
8/18	-	0	0	0	0	0	0	0	-	1	0	0	-	-
Average	125	0	0.1	0	0	0.1	0	0	97	0.2	0	0	0	0

(continued)

TABLE 3. (continued)

Date	8B	9	10	11S	11D	12A	12B	14	SA	SB	WBUS	WBDS	INF	EFF
1/6	0	0	-	-	-	-	0	-	5	4	-	-	-	-
1/16	-	-	-	0	-	-	0	-	15	-	-	-	-	-
1/28	0	0	-	-	-	-	-	-	-	-	-	-	TNTC	TNTC
2/4	-	-	-	1	-	-	-	-	0	11	31	30	-	-
2/13	0	-	0	0	-	-	0	-	4	-	-	-	TNTC	TNTC
2/20	-	-	-	-	-	-	-	-	0	7	48	42	-	-
2/25	0	-	0	0	-	-	-	-	-	-	-	-	308*	230*
3/5	-	-	-	-	-	-	0	-	0	0	1	0	-	-
3/10	-	-	-	-	-	-	-	-	-	-	-	-	540*	380*
3/31	-	0	0	-	-	-	0	-	-	-	-	-	-	-
4/6	0	0	-	0	0	-	-	-	-	-	-	-	500*	170*
4/15	-	-	-	-	-	-	0	-	0	1	1	0	-	-
4/22	-	-	0	0	0	-	-	-	-	-	-	-	400*	390*
4/29	-	0	-	0	0	-	-	-	0	0	4	4	-	-
5/20	-	0	0	-	-	-	0	-	-	-	-	-	400*	-
5/27	-	-	-	-	0	-	0	-	12	0	-	17	-	-
6/16	-	1	-	0	0	0	0	-	7	30	18	21	400*	190*
6/24	-	0	-	-	-	0	0	0	7	8	29	41	-	-
7/17	-	-	-	0	0	0	0	-	-	-	-	-	790*	660*
8/10	-	0	-	-	0	-	-	0	0	169	15	29	-	-
8/18	-	0	-	0	0	0	0	-	0	2	0	0	900*	760*
Average	0	0.1	0	0.1	0	0	0	0	3.9	21	18	18	530*	397*

* Expressed as 1000's per 100 ml.

TABLE 4. ENTEROCOCCI (ml^{-1})

Location	6/24	7/17	8/10	Average
1	3	14	1	5.8
2A	0	0	0	0
2B	0	0	0	0
3A	0	0	0	0
3B	0	1	0	0.17
3C	0	0	0	0
3D	0	0	0	0
4	0	0	0	0
5	-	0	0	0
6A	0	0	0	0
6B	0	0	0	0
7	0	0	0	0
8AA	-	0	-	0
8A	-	-	-	0
8B	-	0	-	-
9	1	0	0	0.17
10	0	0	0	0
11S	0	0	0	0
11D	0	0	0	0
12A	-	0	-	0
12B	-	0	-	0
14	0	-	0	0
SA	1	1	0	0.5
SB	6	1	1	2.3
WBUS	4	3	1	2.7
WBDS	4	3	1	2.7
INF	-	15,500	-	15,500
EFF	-	9,500	-	9,500
Blank	0	0	0	0

of coliphage. The highest values were found in wells 1 and 9 but no pattern seemed to be established for any of the wells. The treatment plant influent and effluent had relatively low concentrations along with the seepage streams and West Brook. Eliassen et al (25) found that viruses were effectively removed in soil systems. Robeck (26) found good removal in only 0.6 m (2 ft) of sand. Apparently, the adsorption capacity for coliphage (and for most virus as well) has been exhausted in the Lake George treatment system; however, the effects of the virus on West Brook (and consequently on Lake George) may not be significant when compared to background levels of the West Brook upstream station. More work needs to be done in this area. Specifically, the sources of contamination in West Brook and the native ground water need to be identified.

TABLE 5. COLIPHAGE (ml^{-1})

Location	6/24	7/17	8/10	Average
1	62	51	87	67
2A	24	37	31	31
2B	30	20	42	31
3A	56	17	31	35
3B	40	36	36	37
3C	51	38	41	43
3D	40	47	60	49
4	27	33	39	33
5	50	27	39	39
6A	10	21	24	18
6B	32	16	34	27
7	31	23	9	21
8AA	-	37	-	37
8A	-	22	-	22
8B	-	-	-	-
9	60	71	43	58
10A	52	42	30	41
11S	26	30	41	32
11D	27	29	45	34
12A	-	20	-	20
12B	-	47	-	47
14	62	-	38	50
SA	6	4	12	7
SB	2	7	8	6
WBUS	12	8	7	9
WBDS	5	11	9	8
INF	15	10	17	14
EFF	11	14	6	10
Blank	0	0	0	0

Three separate BOD_5 studies were conducted in this treatment process evaluation. The first study dealt with influent and effluent BOD_5 of the conventional portion of the treatment plant, and the results are presented in Table 6. The second study dealt with the BOD_5 reduction in the effluent as it stood on the sand beds prior to complete infiltration, while the third study involved determining BOD_5 concentrations at all other stations monitored including wells, seeps, and streams.

TABLE 6. REDUCTION IN BOD IN THE CONVENTIONAL PORTION OF THE LAKE GEORGE VILLAGE SEWAGE TREATMENT PLANT

	BOD_5, mg/l		
Date	Influent	Effluent	% Reduction
1/13 & 1/20/75 (Avg)	35.5	14.8	58
1/29/75	14.5*	14.1	18
1/29/75	19.9**		
5/18/76	-	21.5	-
6/7/76	72.5	13.8	81
6/9/76	86.3	36.3	58
6/12/76	88.8	20.6	77
6/14/76	69.4	35.4	49
6/15/76	61.9	21.4	65
6/16/76	35.6	15.0	58
6/17/76	91.9	20.6	78
8/3/76	210.0	40.3	81
8/5/76	145.0	18.8	87
8/6/76	154.0	20.6	87
8/8/76	199.0	42.5	77
8/9/76	138.0	78.8	43

* Sewer from Lake George Village
** Sewer from Caldwell District (Town of Lake George)

(1975 data ref. [27], 1976 data ref. [23])

In January the BOD_5 concentrations of the plant influent were very low and, as a result, BOD removed in the plant was also low. The average for the January 13 and 20 samples indicated a 54 percent reduction through the trickling filter which was considered satisfactory for a period during which the temperature of the wastewater was about 5°C. During June the influent BOD_5 averaged 72 mg/l, and the effluent averaged 23 mg/l for a 68 percent reduction through the treatment process with the trickling filter alone contributing to 32 percent reduction. In August the influent BOD_5 rose to an average value of 169 mg/l while the effluent BOD_5 was reduced to 40 mg/l indicating a 76 percent reduction in the primary and secondary processes. Even though the percent reduction was high in the conventional treatment process during the summer months, there was a large increase in both the influent and effluent BOD_5 and subsequent high BOD_5 loading of the sand beds. The flows were also higher in August, thus making the total bed loading much higher than other seasons. The relatively low influent BOD_5 in January and June indicates that the sewage was being diluted by sewer line infiltration while the higher flows in August masked this infiltration. If the infiltration could be eliminated, obviously a substantial hydraulic load would be removed from the plant.

In the second study evaluating BOD_5 reduction on the sand beds, bed N-4 was chosen for study because the effluent took longer to infiltrate from this bed than others in the system. Most beds drained after only one or two days; however, effluent remained on the surface of this bed for almost four days. The effluent applied to the bed on May 18, 1976, had a BOD_5 of about 22 mg/l (Table 7). The sample collected from the bed on that day after several hours of flooding had a BOD_5 of about 13 mg/l. This reduction was probably due to settling of particulate matter from the ponded wastewater onto the surface of the sand bed. By the following day, the BOD_5 of the wastewater had dropped to 7 mg/l which was a reduction of 59 percent. This reduction was theorized to be due to combination of additional sedimentation and biological removal. By May 20, the BOD_5 had dropped to 5 mg/l but increased slightly on the following day. This increase may have been due to algal growths in the ponded water or due to wind mixing which would have disturbed the sediments in the shallow water depth. On the last day before the effluent had disappeared into the subsurface, the BOD_5 had again dropped to 5 mg/l. These values indicate a substantial BOD_5 reduction of the wastewater while standing on the infiltration beds. In this test, a 76 percent reduction was experienced during the four days it took for the plant effluent to infiltrate into the sand. As previously mentioned, the effluent infiltrates into the subsurface much more rapidly in other beds, but most of the BOD_5 reduction occurred within only one day of detention.

TABLE 7. BOD_5 REDUCTION WHILE STANDING ON A SLOWLY INFILTRATING SAND BED

Date	BOD_5, mg/l
5/18/76 Applied Effluent	21.5
5/18/76 On Bed	12.8
5/19/76 On Bed	7.3
5/20/76 On Bed	5.0
5/21/76 On Bed	7.8
5/22/76 On Bed	5.3

The results in Table 8 show BOD_5 values collected during the third phase of the study involving wells, seeps, and stream stations. These results indicate low, but measureable, BOD_5 at every location including West Brook and the control wells. In most cases these low BOD_5's were satisfied by sufficient DO in the wells. However, wells 1, 2B, 3D, 5, 6B, 9, and 14 generally did not contain enough DO to satisfy the BOD_5. These wells also experienced relatively high ammonia concentrations which are indicative of anaerobic conditions. The ammonia could have been produced by bacteria which used the oxygen present in nitrates as a substitute for free DO. Ammonia exerts an oxygen demand on the subsurface system and appears as BOD in that test.

HYDRAULIC CHARACTERISTICS OF THE SAND SYSTEM

Bed Dosing

The flow of the treatment plant influent is gauged by two Parshall flumes. One of the flumes records the flow from the Caldwell Sewer District (Town of Lake George) and the other records the flow from the Village of Lake George. These flows are combined, along with flow from the trickling filter recycle line, prior to primary sedimentation. Daily readings from the totalizer meter are recorded by plant personnel at approximately 8:00am. Rather than use conventional monthly flows, seasonal months were set up since they better correspond to population variations. These seasonal months start on the 21st of the previous month and run through the 20th of the stated month. The mean monthly flows are presented in Table 9 in units of liters per second (1 mgd = 43.803 liters per second). Total seasonal flows were previously shown in Figure 5. The steady increase in peak summer flows during the study period was accompanied by an even greater increase in minimum winter flows. While August flows increased 45 percent from 1969 to 1976, January flows increased 129 percent during the same time span. The main reason for the increase in winter flows was the increase in winter tourism due to the rising popularity of skiing, snowmobiling, and other winter sports, plus a growth in attendance at the annual Lake George Winter Festival, normally held every weekend in February. Based on earlier data, it would appear that the maximum flow in the summer of 1976 should have exceeded 50 l/sec, but the lower value can be explained by the relatively cold wet weather encountered that year which reduced tourism. In contrast, the summer of 1975 was hot and dry, and the large tourist population accounted for extremely high flows during that summer.

The treatment plant was originally designed to treat 21.9 l/sec (0.5 mgd), but was subsequently expanded to a stated capacity of 76.6 l/sec (1.75 mgd). Currently peak daily flows are as high as 55 l/sec (1.25 mgd) in the summer. A major problem in plant operation is caused by flow surges resulting from the intermittent sewage pumping to the plant. The high rate pump from the Caldwell Sewer District operates at 220 l/sec (5 mgd). Even if flows from the Village and recycle line were absent, this high pumping rate would disrupt the system. When sewage is pumped from both the Town and the Village, the influent flow to the primary settling tank is so large that it stirs up settled solids upsetting both the trickling filters and the secondary settling tanks. This results in an effluent whose BOD is sometimes higher than the simultaneous influent, which leads to rapid sand bed clogging.

TABLE 8. BIOCHEMICAL OXYGEN DEMAND (5-day), mg/l

Sample	Date	BOD_5	DO
1	6/24	2.4	2.5
1	8/3	0.7	0.7
2A	6/24	2.6	5.7
2A	7/17	2.0	6.0
2A	8/3	1.9	6.0
2B	6/24	2.1	1.3
2B	7/17	1.5	1.7
2B	8/3	0.9	1.7
3A	6/24	0.8	1.6
3B	6/24	0.7	0.9
3C	6/24	0.6	0.8
3D	6/24	1.1	0.9
4	6/24	1.2	1.2
5	6/24	2.0	1.1
5	8/3	1.9	1.1
6A	6/24	2.5	6.1
6A	7/17	1.2	4.6
6A	8/3	0.8	4.6
6B	6/24	2.7	0.9
6B	7/17	1.0	1.0
6B	8/3	1.7	1.0
7	8/3	0.6	2.0
9	6/24	3.1	1.1
9	7/17	1.6	0.5
9	8/3	2.2	0.5
10	6/24	0.5	3.4
11S	7/17	0.8	6.8
11D	7/17	1.2	7.1
12A	7/17	1.8	5.0
12B	7/17	1.4	2.0
14	6/24	4.0	1.0
14	7/17	1.7	1.1
14	8/3	1.1	1.1
SA	6/24	1.4	8.4
SA	7/17	0.6	7.4
SA	8/3	1.6	7.4
SB	6/24	1.6	7.0
SB	7/17	1.0	6.0
SB	8/3	0.3	6.0
WBUS	6/24	1.6	10.4
WBUS	7/17	0.4	8.1
WBUS	8/3	0.9	8.1
WBDS	6/24	1.5	8.5
WBDS	7/17	0.4	7.6
WBDS	8/3	0.9	7.6

Note: All dissolved oxygen (DO) values are expressed as mg/l.

TABLE 9. MEAN MONTHLY FLOWS (liters/second)

Month	1968	1969	1970	1971	1972	1973	1974	1975	1976
Jan	-	9.46	14.06	14.81	17.22	17.17	25.01	21.86	21.64
Feb	-	8.59	16.38	15.20	17.44	19.89	22.04	22.47	27.47
Mar	-	11.96	14.89	16.21	19.19	21.03	25.76	24.88	32.72
Apr	-	16.43	20.55	22.74	24.93	24.18	28.26	30.62	32.59
May	-	21.86	20.46	23.74	29.31	24.97	28.52	28.96	32.24
Jun	-	21.60	24.14	26.37	29.04	29.66	30.67	34.13	34.87
Jul	25.45	27.99	33.16	36.89	41.22	39.25	40.22	42.49	41.97
Aug	31.72	36.23	33.16	44.16	42.41	43.68	43.15	49.42	45.91
Sep	21.68	25.50	29.35	37.54	29.70	33.95	32.42	35.75	34.96
Oct	13.67	14.94	19.49	23.61	19.19	22.74	25.76	32.86	-
Nov	11.57	13.14	14.63	16.82	28.39	18.05	18.14	25.76	-
Dec	10.73	13.19	13.76	15.86	18.53	18.88	21.51	23.44	-

The effective infiltration areas of the individual sand beds were measured by Beyer (10) in 1975, as shown in Table 10. North beds 13 and 14 were treated as one bed by plant personnel since they are always dosed simultaneously; hence they are combined in Table 10. Apparently little attention was paid to constructing the initial six and subsequent eight north sand beds of equal area; however, the south beds were constructed much more uniformly. The total area actually in use for infiltration is somewhat less than previous estimates (9). Apparently previous estimates included the separating dikes as well as the infiltration beds.

The dosing records kept by plant personnel were used to determine the frequency of bed use for each season. Table 11 shows the number of times each bed was dosed per season for four years for which data were available. The frequency with which any one sand bed was dosed was determined by a daily evaluation by plant personnel of the condition of the beds and the need for beds. Obviously, those beds which drained and dried fastest were dosed more frequently. Bed selection was also based on consideration of where the location of the distribution chambers were. If, for example, both beds S-1 and S-5 were available for dosing, bed S-1 would probably have been dosed because only one gate at one distribution chamber would have had to be opened. To dose bed S-5, three gates at three distribution chambers have to be opened. Since the normal procedure was to dose one north and one south bed simultaneously, and since there are fewer south beds, the south beds are dosed more frequently. The south beds are also newer and drain more rapidly than the north beds.

In calculating the depth of effluent applied to each bed, it was assumed that one half the daily flow occurred between the hours of 8:00am and 4:00pm. If four beds were dosed daily with the dosing changed at 8:00am and 4:00pm, which was the normal operating procedure, one fourth the daily flow was applied to each bed. If three beds were dosed between 8:00am and 4:00pm and two beds were dosed from 4:00pm to 8:00am, the first three beds each received 1/6 the daily flow (1/3 x 1/2) and the second two beds each received 1/4 the daily flow (1/2 x 1/2). Using similar reasoning, the daily flows to each bed were calculated using the plant flow and bed dosing records. These daily bed loadings were then totalled for each bed by season. Finally, using the surface area of each bed, the depth of effluent applied to each bed during each season was determined as presented in Table 12.

Bed S-1 received a significantly higher loading rate than any other bed, indicating experienced plant personnel were utilizing its high infiltration rate. Using similar reasoning, after review of the data, it might be theorized that beds N-7, N-13-14, N-8, and N-5 had the lowest infiltration rates. However, this reasoning is not entirely valid since some beds which were available for dosing were not always needed and were only flooded when flows were high enough to require their use. During peak flow periods, it often became necessary to dose beds which still contained effluent from previous dosings. Also, the time required for cleaning each bed was highly variant which resulted in different lengths of time each bed was out of service. Therefore, realizing the non-uniformity of bed dosing operations, the loadings presented in Table 12 should be considered as only approximations for estimating the infiltration rates of each individual bed.

TABLE 10. SAND BED AREAS

Sand Bed	Area	
	m^2	ft^2
N-1	908.0	9,774
N-2	986.0	10,613
N-3	853.0	9,178
N-4	1037.0	11,160
N-5	1185.0	12,754
N-6	955.0	10,284
N-7	1730.0	18,619
N-8	1062.0	11,427
N-9	964.0	10,373
N-10	809.0	8,704
N-11	1361.0	14,646
N-12	1154.0	12,425
N-13-14	1500.0	16,151
S-1	1118.0	12,035
S-2	1118.0	12,035
S-3	1135.0	12,218
S-4	1135.0	12,218
S-5	1193.0	12,846
S-6	1200.0	12,915
S-7	1234.0	13,248
TOTAL NORTH BEDS	14502.0	146,105
TOTAL SOUTH BEDS	8134.0	87,551
TOTAL	22636.0	233,656

TABLE 11. BED DOSING FREQUENCY

		1973				1974			1975			1976		
Bed	S	F	W	S	S	F	W	S	S	F	W	S	S	Total
N-1	1	14	13	10	9	13	10	9	9	20	15	12	7	142
N-2	1	15	15	9	12	12	14	11	7	13	12	18	16	155
N-3	2	11	6	16	15	11	7	11	11	13	13	10	18	144
N-4	4	13	8	12	18	14	12	12	11	13	15	13	15	160
N-5	0	12	8	7	13	10	10	9	11	12	8	9	14	123
N-6	5	15	16	23	21	18	10	16	17	19	9	25	22	216
N-7	0	6	9	10	10	6	9	8	8	10	8	9	9	102
N-8	2	7	7	10	10	7	6	9	11	9	9	9	10	106
N-9	7	19	24	25	28	18	29	22	13	15	15	14	18	247
N-10	5	13	18	16	23	15	16	12	12	10	14	12	16	182
N-11	4	17	13	11	25	19	12	22	22	12	13	15	14	199
N-12	11	29	33	37	42	25	33	30	29	14	19	15	16	333
N-13-14	1	13	11	17	17	15	12	13	13	12	11	10	10	155
S-1	6	37	38	45	51	28	39	42	27	25	31	29	32	430
S-2	7	12	26	19	21	25	0	0	19	26	20	20	29	224
S-3	2	30	23	28	38	17	31	28	26	15	28	26	25	317
S-4	6	24	19	25	25	31	27	30	26	24	12	28	25	302
S-5	8	25	22	28	39	29	30	26	24	22	22	9	29	313
S-6	11	38	23	26	42	32	38	35	28	21	23	26	26	369
S-7	6	20	28	26	28	22	15	25	19	21	21	22	14	267

TABLE 12. DEPTH OF APPLIED SEWAGE (meters)

Bed	1975 Fall	1976 Winter	1976 Spring	1976 Summer	Total
N-1	12.32	9.47	9.68	7.98	39.45
N-2	7.31	6.82	12.77	12.03	38.93
N-3	8.19	9.20	7.69	17.51	42.59
N-4	7.23	8.22	8.73	12.76	36.94
N-5	5.49	3.59	5.24	8.66	22.98
N-6	11.24	5.21	18.60	18.64	53.69
N-7	3.95	2.63	3.37	4.06	14.01
N-8	4.36	4.54	5.83	6.96	21.69
N-9	10.73	10.77	12.48	16.93	50.91
N-10	11.91	13.02	11.89	16.93	53.75
N-11	7.46	8.49	11.06	12.04	39.05
N-12	12.52	15.44	15.05	16.26	59.27
N-13-14	5.85	4.08	6.37	5.09	21.39
S-1	17.37	21.66	22.91	27.58	89.52
S-2	13.42	10.73	12.98	21.54	58.67
S-3	9.26	18.79	19.36	22.16	69.57
S-4	11.86	5.28	18.46	18.07	53.67
S-5	12.43	11.42	5.99	24.05	53.89
S-6	11.81	12.30	18.53	20.03	62.67
S-7	11.28	10.43	13.17	8.27	43.15

Total monthly precipitation, calculated from data collected in a precipitation gauge located at the treatment plant, is presented in Table 13 along with the monthly averages of depth of sewage plant effluent applied to each sand bed during one complete year of study. The last column of Table 13 shows the percent of precipitation effecting the total hydraulic loading. The yearly average contribution of precipitation to the bed loading was 2.59 percent. This average contribution was insignificant to the total volume infiltrated; however, in August 1976, a heavy rainfall occurred during a period of peak flow. This incident not only added an additional hydraulic load to the plant, but also disrupted cleaning operations. Although precipitation may not have a significant direct effect on bed loadings, it may indirectly affect the plant hydraulics and operation.

TABLE 13. PRECIPITATION EFFECTS

Month	Depth of Applied Sewage (meters)	Precipitation (cm)	Percent - Precip. of Total
9/75	4.330	12.70	2.85
10/75	3.833	13.16	3.33
11/75	3.206	7.57	2.32
12/75	2.761	4.52	1.60
1/76	2.631	9.22	3.38
2/76	3.348	8.76	2.56
3/76	3.626	10.49	2.81
4/76	3.969	6.93	1.71
5/76	3.876	20.27	4.98
6/76	4.163	5.97	1.42
7/76	5.012	10.92	2.13
8/76	5.646	11.43	1.98

During periods of high flows, the primary and secondary treatment processes operate very inefficiently, resulting in a high carryover of solids to the sand beds, which decreases the infiltration rate. Although the beds should be cleaned frequently, they cannot be taken out of service for long periods during high flow periods. Bed cleaning is normally slowed during summer months due to additional time required for weed removal in the beds. Severe conditions occurred during August 1975 and 1976 requiring sewage effluent to be pumped to adjacent forest lands for periods exceeding two weeks.

Winter operation is of primary interest in this study since the site is located in the cold upper New York State climate. Within the primary treatment system, the only modifications necessary during sub-zero winter temperatures and heavy snow are covering the Parshall flumes and the 2-compartment settling tanks with boards. Secondary treatment is modified during winter by diverting the flow from the rotary high rate trickling filters to the standard rate fixed nozzle sprinkling filter which is covered with boards. Temperatures are maintained above freezing at all times within the covered trickling filter. Heavy snow cover provides insulation retaining heat from the wastewater.

Primary interest is the operational kinetics of the sand beds during the winter. The treatment system is operated continuously through the harsh winters without effluent storage. Normal dosing continues using two beds per cycle. Reed et al (28) have indicated that an infiltration-percolation system could not work in the harsh climate of the northeast. He suggested that freezing of the moist soil surface would create an impervious layer halting percolation of effluent. Figure 42 shows this did not occur in the Lake George System. The first photo shows the initial dosing of an ice and snow covered bed. The ice and snow near the splash pad melted very quickly during flooding. From this point the warm effluent flowed under the ice simultaneously melting the ice above it and the ground below it. The second photo indicates continuous melting as the depth of effluent in the bed increased. Within a few hours all that remained of the ice in the beds was a few floating pieces that eventually melted. Infiltration proceeded, as in other seasons, with the ambient temperature cooling the effluent standing on the bed. By the time the ponded effluent had cooled and begun freezing again, the majority of the effluent had entered the subsurface. An ice layer formed in this manner may be beneficial, serving as an insulating layer for the soil surface.

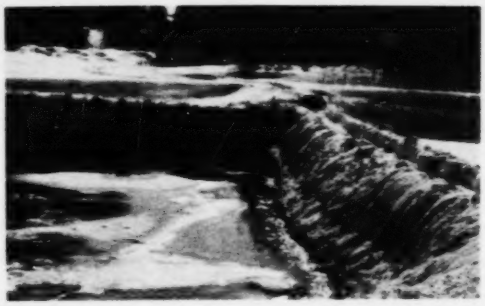
Infiltration Studies

The principal advantage of the infiltration-percolation method of land treatment is that the application rate is greater per unit area than with other land treatment systems. Studies were made to establish the approximate infiltration rates and to determine the dynamics of infiltration existing at the Lake George Village Sewage Treatment Plant.

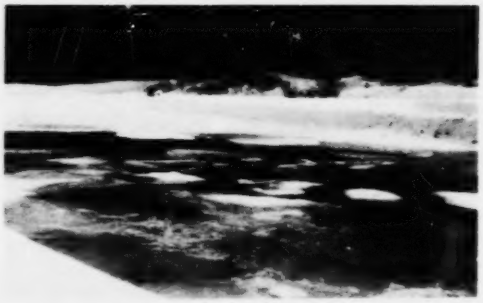
The first infiltration rate studies were attempted using a staff gauge mounted in a sand bed from which periodic water level readings were recorded. This method failed to give an accurate view of the process, since insufficient readings were taken. Results from the recording gauge in the seepage above Gage Road Station indicated a virtually constant flow in the seepage stream. Therefore, this Stevens Type F recording gauge was removed from the seepage above Gage Road Station and installed in a sand bed. A second gauge was installed in another bed under identical conditions. Stilling wells were not necessary when using the Type F gauge in the sand beds since the beds were recessed below ground level and wave action due to wind was minimal.

The sand beds used to study infiltration rates were selected on the basis of availability. Work schedules of the treatment plant personnel with respect to cleaning and dosing were often the determining factors in choosing a bed for study during a specific time, particularly since normal operational procedures of the facility included a drying period between dosings.

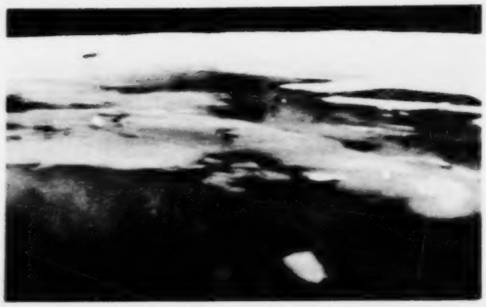
One of the major problems in measuring accurate infiltration rates was the inability to determine the initial rate during dosing. Reed et al. (28) stated that the greatest infiltration rate occurred at the time of initial soil wetting.



Dosing an ice and snow covered bed



Melting of ice and snow as the bed fills with treated sewage



Melting of ice and snow almost complete within a few hours

FIGURE 42. Winter Operation of a Sand Bed
Bed S-4 - February 7, 1975.

Similar results were observed at Lake George. For example, bed N-9 dosed on August 20, 1975, to a depth of 0.15 m (0.5 ft) recorded an extremely high infiltration rate of 0.9 m/day (2.94 ft/day). In all probability this infiltration rate was even higher at the beginning of bed dosing. However, infiltration measurements in this study were begun on each bed at the time flooding stopped. Thus, these measurements reflect a more accurate evaluation of the actual uniform infiltration rate during continuous operation.

A general comparison of infiltration rates of several sand beds at corresponding heads of effluent is shown in Figure 43. These rates represent normal operation which is considered to be the operational period following a week of flooding cycles after the beds had been cleaned and scraped. The infiltration rate appeared to be a function of head after the initial soil wetting. This function tended to be invalid at lower heads, especially in the older sand beds. The oldest north beds generally had slower infiltration rates than the newer north beds; however, the highest rates in all beds were generally found in the south beds.

A study was made to determine if the infiltration rate declined with bed use and to what extent it was renewed with a drying period between dosings. The beds examined were N-5 and N-11. Bed N-11 had drying periods of approximately 24 hours between draining and dosing while bed N-5 had a 12 hour drying period. Longer drying periods would have been desirable but the summer months at Lake George do not allow the plant operators this luxury. Even with these short recuperative periods, there was little reduction in infiltration rates as shown in Figure 44.

Figure 45 illustrates the effect of bed cleaning in increasing the infiltration rate. This Figure also shows a decline in the infiltration rate after successive dosings. The initial rates recorded exceeded 0.6 m/day (2 ft/day), and higher rates would probably have been measured if recording equipment would have been operating properly during initial bed filling. It was anticipated that infiltration in bed S-5 would have continued to decrease during the next few dosings until it approached the curve of bed S-7, which is shown to illustrate normal conditions without recent bed cleaning.

These studies were followed by an evaluation of the infiltration rate when beds are continuously flooded after cleaning. As can be observed in Figure 46, the response of the bed was similar to that of a bed operated with drying periods between dosings. There was a general reduction in infiltration rate with time after each successive dosing when the bed was refilled while still inundated from the previous dosing. The rate quickly decreased to a rate approaching that of a normally operated bed. The curve of bed N-5 was included to compare this study with normal operation where beds are rested between dosings.

The treatment plant was plagued during the summer of 1975 with weed growths in the beds. The major problem caused by the weeds was that they interferred with raking the beds to remove the organic mat on the bed surface. This occurrence presented an opportunity to evaluate the effect of weed growth in the beds on the infiltration rate. The infiltration rate was found to be very low when measured

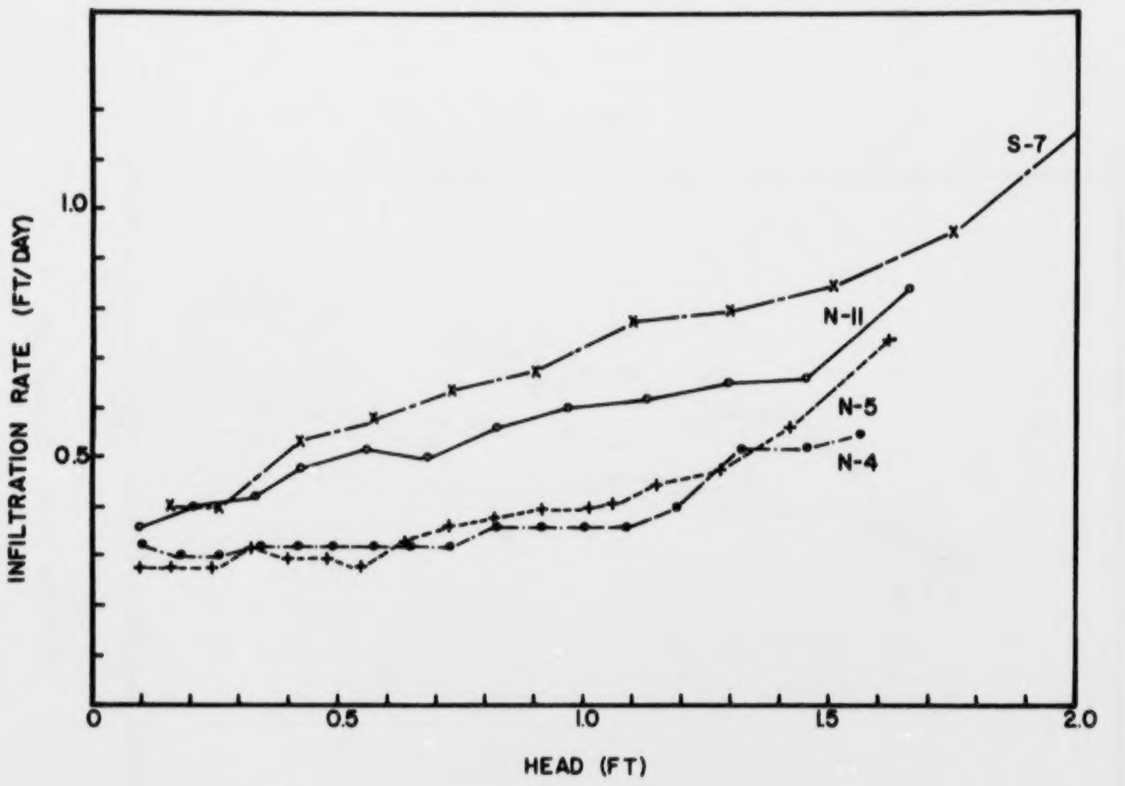


Figure 43. Infiltration Rates of Several Sand Beds During Normal Operation.

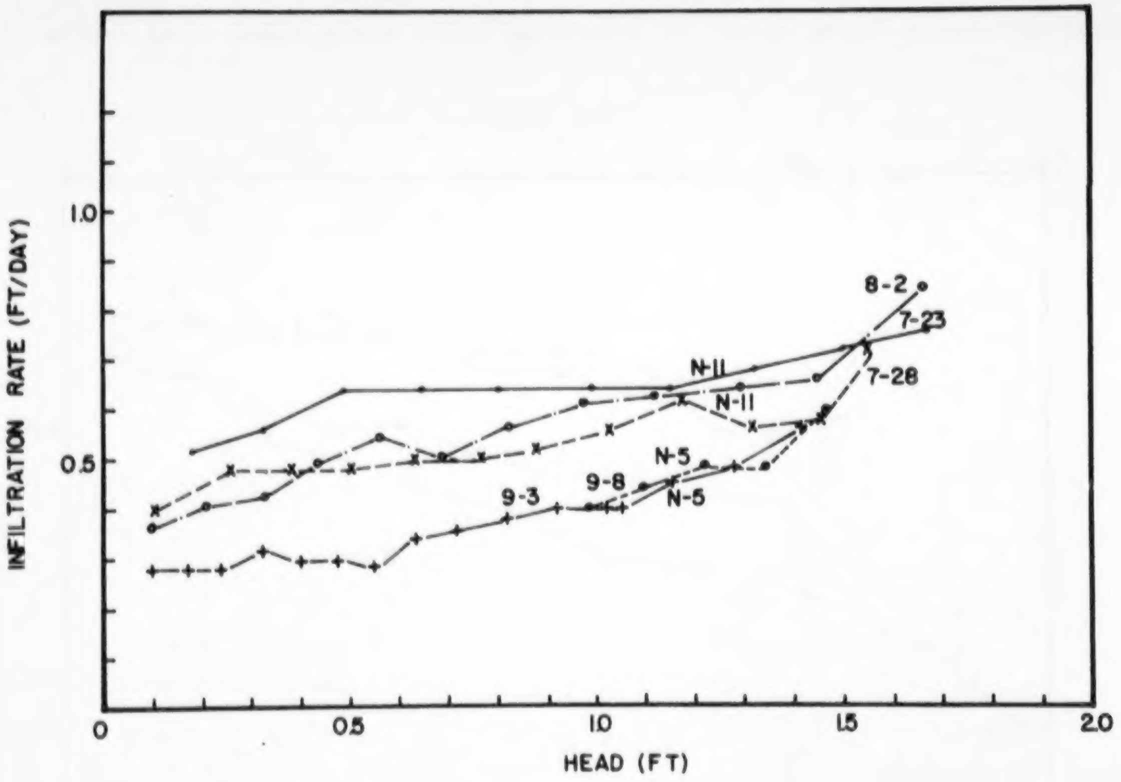


Figure 44. Changes in Infiltration Rates of Several Sand Beds Under Normal Dosing Operations Including Normal Drying Periods.

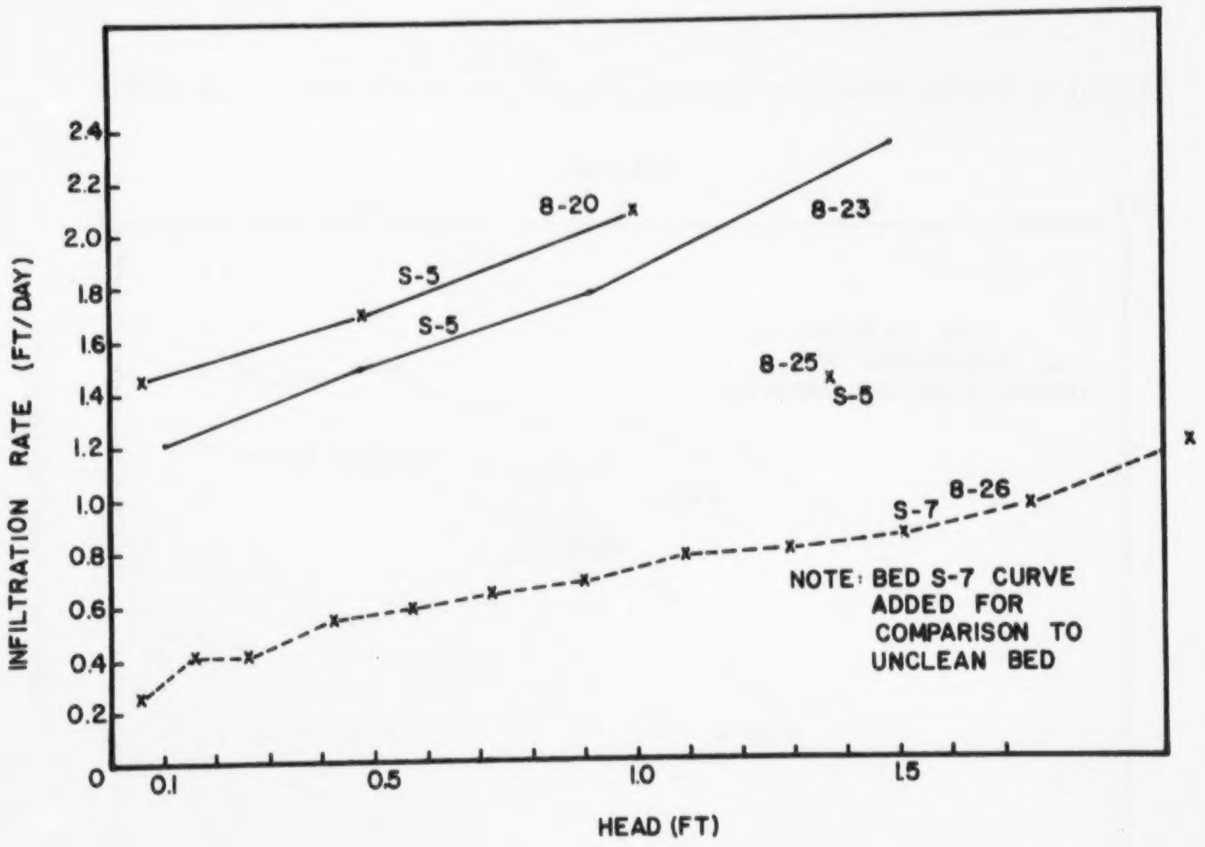


Figure 45. Infiltration Rate in Bed S-5 Showing the Decline in Rate With Continued Normal Operation after Cleaning the Bed.

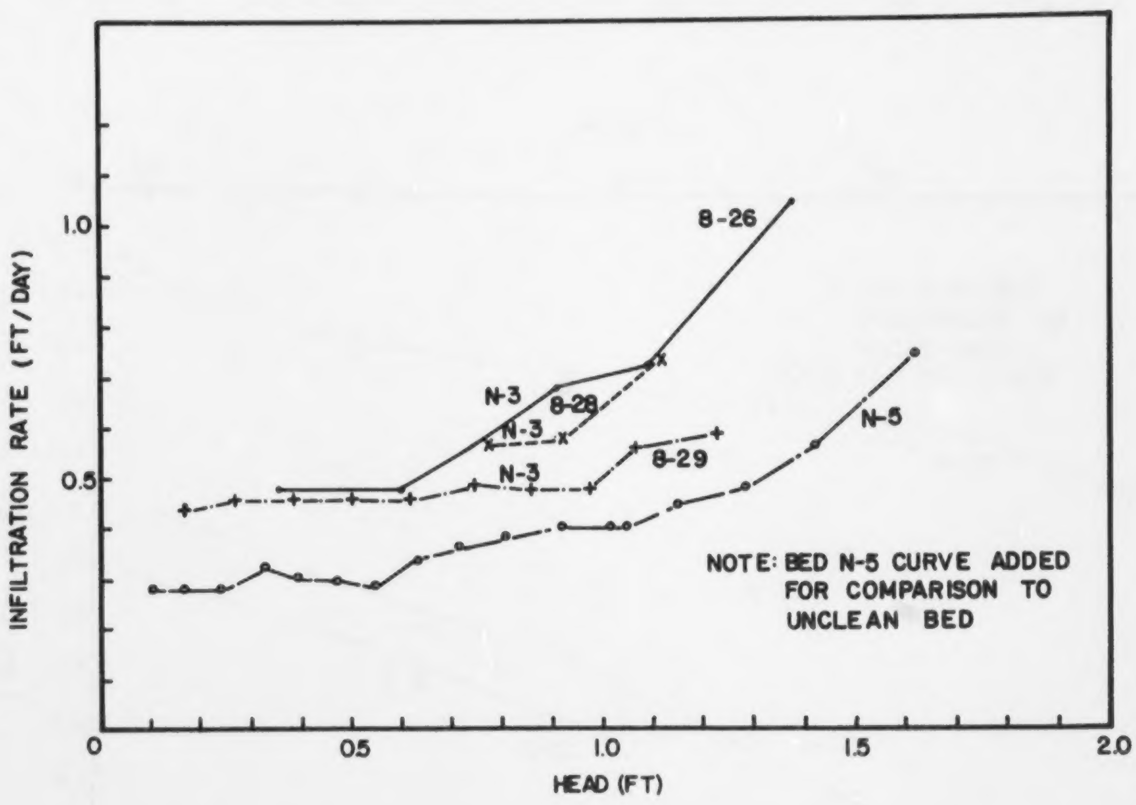


Figure 46. Infiltration Rate in Bed N-3 with Continuous Dosing Following Cleaning of the Bed.

in weed infested beds with shallow depths of wastewater. These low rates resulted in the beds having to be flooded many times during the summer while they contained shallow depths of effluents. The results of this unusual operation was compared with the operation of a bed located nearby which was maintained without vegetation, yet under continuous inundation. The results of this experiment are shown in Figure 47. Head appeared to have a greater influence on the infiltration rate in a weed covered bed than a clean bed. In the weed covered bed, at high heads (0.5 m), the rate approached 0.3 m/day (1 ft/day), but at low or average heads 0.3 m), the rate was less than 0.15 m/day (0.5 ft/day). The dosings of S-2 (bare surface on bed) and S-3 (vegetation on surface) were approximately the same. The data from S-2 were similar to data from other bare surfaced beds, but the data collected from S-3 were unique and must be attributed to the vegetative cover. The curve shown in Figure 47 of bed S-3 exhibited a much steeper slope indicating a significant reduction of the infiltration rate at low heads and an increased infiltration rate at high heads. It also should be noted that neither bed exhibited any significant reduction in rate during continuous inundation.

At the average loading depth of 0.3 m (1 ft), the average infiltration rate for beds under normal conditions was greater than 0.15 m/day (0.5 ft/day). The bed storage capacity with this infiltration rate has been adequate to handle the sewage effluent discharges with the possible exception of August. Under normal conditions, the infiltration rates were higher at higher heads indicating the dependence of rate on head, particularly at high head conditions. Higher infiltration rates occurred after cleaning the beds and, consequently, frequent cleaning of the sand beds increased the loading capacity of the system. The slopes of the curves indicated head to be a greater factor in a freshly cleaned bed than one where matting was allowed to form on the bed surface. The operation of bed S-2 under continuous inundation did not show a continuous reduction in rate, but the infiltration rates measured for this bed were significantly less than the rates of other south beds operated under normal wetting and drying cycles. The use of bed S-2 to hold the sludge from the primary settling tank during the winter of 1974-75 could have affected its infiltration capacity and masked the true infiltration rates that can be expected when operating with beds continuously inundated.

Percolation Rates

Information concerning the rate of flow of water within the sand is needed to determine the maximum effluent application rates, any buildup of water within the sand, and the time of contact in both the unsaturated and saturated sand zones. Tracers were used to determine percolation rates and direction of flow to evaluate sampling site locations for adequate data collection. Tracer techniques used involved the addition of substances not normally present in either the natural ground water or the applied wastewater, or if present, these substances were added in concentrations significantly higher than existing so that positive identification of tracer movement could be made. Samples were collected at various locations in a systematic manner in order that calculations of the time of vertical and horizontal travel of tracers and applied effluent could be determined between sample locations.

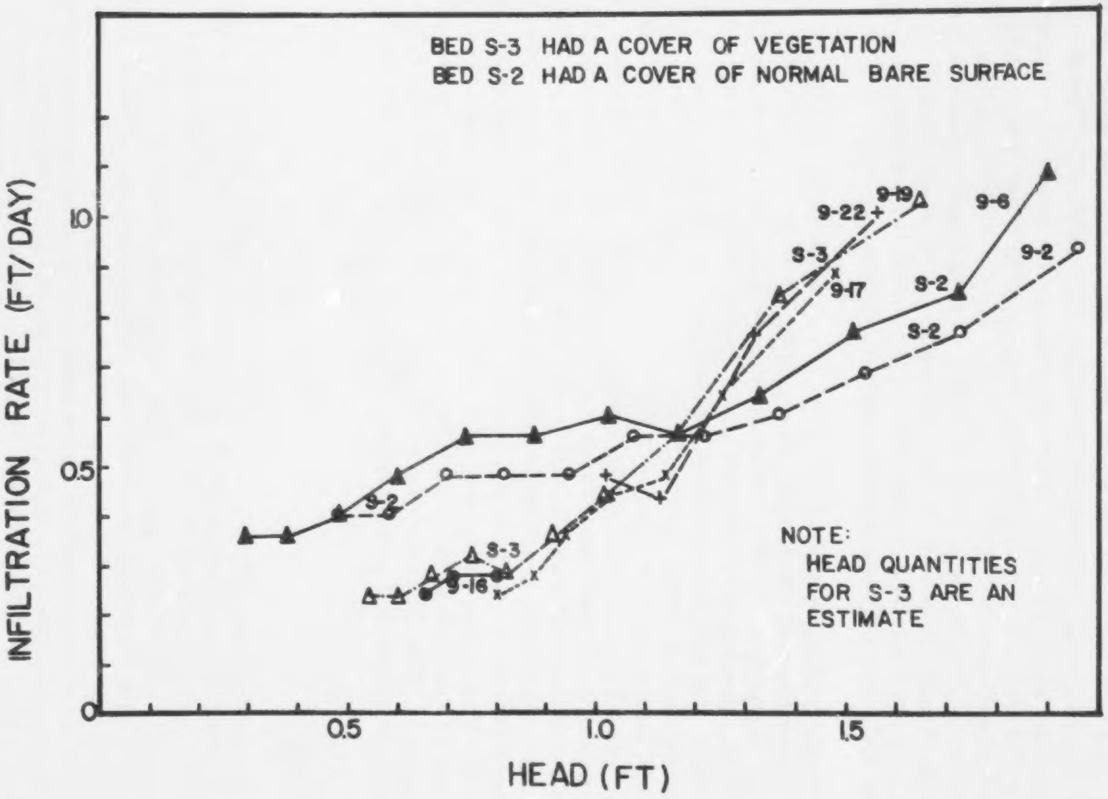


Figure 47. Comparison of Two Beds Under Constant Inundation One with a Weed Cover and One with the Weeds Removed.

Vertical Flow Through the Unsaturated Sand --

After any applied effluent infiltrates into a sand, it first percolates downward through the unsaturated sand in essentially a vertical direction until it reaches a saturated zone. As wastewater reaches the saturated zone, it becomes dispersed by diffusion and moves along with the native water in the saturated zone generally at the same velocity and direction as the native water. The first studies to determine the vertical percolation rate were conducted by Glavin and Romero-Rojas in 1968 (29). Using rhodamine B dye, they established a vertical percolation rate of approximately 8 m/day (27 ft/day) in the top 1.5m (5 ft) of sand beds N-11 and N-13. This value appears extremely high and after comparing the results with more recent work, it seems likely that the applied effluent in which the dye was placed was moving down the outside of a poorly sealed well pipe to the well screen, giving erroneous results.

All the more recent tracer studies were performed in bed N-11 utilizing well points, lysimeters, and sampling wells equipped with submersible pumps as shown in Figures 9 and 10. The first attempted dye studies for this project began March 26, 1975, using rhodamine B dye. The dye was added at 10:45am after 2 hours and 45 minutes of bed dosing. Mixing of the dye was accomplished by stirring the ponded water by wading and by the natural wind action. No usable data were collected due to equipment failure.

The second attempt to monitor dye movement through the sand beds was performed during the month of June 1975 using rhodamine WT which is less subject to adsorption onto sand particles than rhodamine B. Bed N-11 was dosed with sewage effluent from 10:10 am June 2 through 8:00 am, June 3. The dye was added at 4:00 pm on June 2, and all the lysimeters were evacuated that day. Figure 48 is the diagrammatic result of this experiment. The effluent moved very rapidly into the soil column, and dye was observed in well point 11-A by 9:00 am on June 3 in very strong concentrations. In the afternoon of June 3, the 3 m (10 ft) deep lysimeter was found to contain dye that registered as a maximum reading on the fluorometer. Sampling continued each day for the first week and then each Monday, Wednesday, and Friday until completion. Two days were found to be the minimum time required for the lysimeters to fill. Well 11-S was pumped continuously during the study and, each night, a flow through flourometer was attached to the well 11-S pump discharge line for continuous monitoring. The peak dye concentration appeared at the 7.6 m (25 ft) depth on June 11. The first dye appeared at the 10.6 m (35 ft) depth 22 days after the study began. The first appearance of dye in the ground water from well 11-S was observed on June 27, 25 days after its addition to the flooded bed. The dye had traveled through approximately 3 m (10 ft) of ground water to enter the well screen in well 11-S. Pump drawdown may have had some effect in pulling the dye through the ground water, but the effect was localized due to the low pumping rate.

The peak concentration of dye reached the 18 m (60 ft) lysimeter 29 days after the study began. The curve in Figure 48 connecting the 3 m, 7.6 m, and 18 m lysimeters is essentially a straight line indicating uniform vertical flow through the unsaturated zone. The peak dye concentration reached well 11-S in the saturated zone during the night of July 3-4, 31 days after the study began.

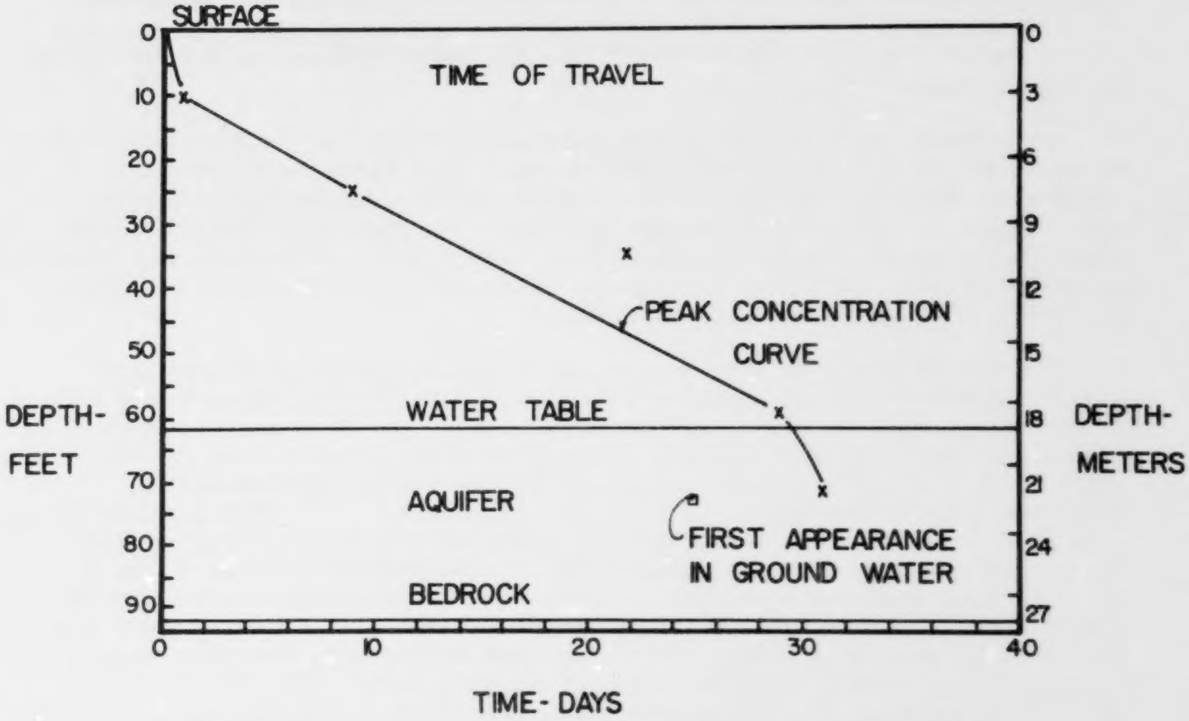


Figure 48. Vertical Transport of Rhodamine WT in Bed N-11 Showing the Peak Concentration Curve and the First Appearance of the Dye in the Ground Water.

The average velocity based on the movement of the peak concentration from the surface to well 11-S was 0.7 m/day (2.32 ft/day). This is about 0.15 m/day (0.5 ft/day) less than the velocity calculated based on the first appearance of dye in the ground water. The velocity through the unsaturated zone between the surface and the 18 m (60 ft) depth was 0.55 m/day (1.79 ft/day). No dye was ever recovered from well 11-D near the bottom of the aquifer. This well was pumped only for a few minutes during each sample collection period.

The dye study was designed to determine: 1) the vertical flow rate of sewage effluent through the unsaturated zone of sand below the infiltration beds; 2) the depth of penetration of sewage effluent in the saturated zone and, 3) the plume configuration and movement within the aquifer. It was hoped that the plume could be followed downgradient, and its changes characterized by sampling the network of test wells between the infiltration beds and seeps entering West Brook. Had the latter portion of this study been successful, it would have confirmed that the seepage areas were the result of ground water recharge from the upgradient sewage infiltration basins. Unfortunately, either the dye was diluted and/or absorbed upon entering the saturated zone, or the observation wells were not located in the plume from bed N-11 because the dye could not be traced in the ground water.

Horizontal Flow in the Saturated Zone--

The previously discussed rhodamine WT tracer studies were unsuccessful in measuring the horizontal movement of sewage; therefore, separate tracer studies were conducted to determine the horizontal transport of infiltrated sewage within the saturated portion of the aquifer below the sand beds. It was concluded after careful consideration that tritium would be the most useful tracer to measure movement within the saturated zone since it essentially is the same as water. It can be detected at very low concentrations so its addition would not affect either ground water or surface water supplies into which it ultimately might flow. Initially it was planned that a 1 curie (Ci) dose of tritium would be used for each of 3 separate tests; however, permission could not be obtained from the New York Department of Environmental Conservation, Bureau of Radiation, for doses this large. Permission was given for use of 0.1 Ci doses for the three separate studies. An estimate was made of the anticipated concentration of the tritium in the ground water after dilution with one day of dosing of sewage effluent onto a sand bed. The counting rates of samples collected from the dosed bed and samples of that water upon arrival at the seepage areas were required to be significantly higher than background tritium in the native ground water if valid flow rate determinations were to be made from the studies. These estimated calculations are as follows:

Flooding of bed N-4 on June 30, 1976 = 0.24 million gallons = 0.89 megaliters

Amount of tritium introduced in the dosing = 0.1 Curie

0.1 Ci/0.89 megaliters = 0.11 Ci/megaliter

(0.11 Ci/megaliter) (1 megaliter/10^6 liters) (10^6 μ Ci/Ci) =

0.11 μ Ci/l in the applied effluent where μ Ci = micro Ci = 10^{-6} Ci.

The combined flow of the seepages at West Brook above and below Gage Road was approximately 3.785 megaliters/day (1 mgd). Thus, the tritium concentration in the combined seepage should have been:

(0.11 μ Ci/l) (0.89 megaliters)/3.785 megaliters = 0.03 μ Ci/l
(0.03 μ Ci/l) (10^{6} pCi/μ Ci) (2.2 dpm/pCi) (1 liter/10^{3} ml) =
66.0 dpm/ml in the combined seepage where:
μCi = 10^{-6} curies
pCi = pico curies = 10^{-12} curies
dpm = disintergrations per minute

A 7 ml sample of water from the seepage area should produce 462 dpm. Based on a counting efficiency of 39.4 percent for the Tri-Carb counting system used, a count of 182 counts per minute (cpm) above background could be expected. Counts of this magnitude are considered adequate for detection. The Beckman liquid scintillation counting system with a counting efficiency of 56.4 percent should yield counts of about 260 cpm above background. The flow in West Brook averages 37.85 megaliters/da (10 mgd) providing an additional 1: 10 dilution of the seepages. Thus, the count above background for West Brook should be approximately 18.2 and 26.0 cpm for the Tri-Carb and Beckman counters, respectively. These values are considered high enough to provide statistically significant results. In all of the above calculations, it was assumed that complete mixing of the introduced tritium in the applied effluent, in the ground water, in the seepages, and in West Brook was obtained. If the tritium was not well mixed in the applied effluent and infiltrated into the ground water as a relatively small plug flow, the "hot front end" could give counts significantly higher than calculated, but detection with wide spread wells would be much more difficult.

Rhodamine WT dye was also considered as a potential tracer. This tracer has the advantages of being detectable in low levels by use of a fluorometer and being visually observable at higher concentrations. At sufficient concentrations the presence of the dye could be observed in the field as samples are collected from the observation wells. The significant disadvantage of using rhodamine WT as a tracer is that it is readily absorbed on soil particles, particularly on clay. Since there is very little clay in the Lake George sand filtration system, rhodamine WT was used in all three tracer studies as a secondary tracer to provide a potential visual observation of the presence of the applied effluent at the numerous sampling stations.

Normally sodium chloride provides a satisfactory tracer for monitoring the flow of ground water. Unfortunately, due to contamination by highway deicing salt stored at the town highway garage located near the sewage treatment system, sodium chloride could not be utilized as a tracer with the accuracy desired. However, due to the lack of a better tracer, 108 kg (240 lb) of NaCl was added to the second tritium tracer study to provide a tertiary tracer to differentiate between the second study and any other tritium tracer study. The amount added was calculated to double the amount of chloride normally present in the applied sewage, and it was also significantly higher than the chlorides in the ground water near the deicing salt storage area.

Potassium, in the form of potassium chloride (KCl), has also been used as a secondary tracer and can be measured in low concentrations. In the third tritium tracer study, 22.7 kg of KCl (50 lbs) was added to differentiate this study from previous studies. The amount added was calculated to double the concentration of potassium normally found in the applied sewage effluent.

In each of the separate tracer studies, samples were secured on a daily basis from all of the downgradient observation wells, the seepage, and West Brook beginning from the time of the tracer injection. During the third tracer study, the sampling frequency was decreased to a biweekly schedule after the first two months due to financial constraints. This schedule was considered satisfactory since, as the duration of the study increased, the tracer would become more dispersed, and it would require more than one day to pass an observation well. This fact was confirmed by earlier results from the third tracer study. Prior to introducing the tritium, background counts were taken on all well points between the point of application and West Brook. Each well point was bailed dry and was allowed to refill for each sample collection assuring fresh aquifer samples.

In the first tracer study, rhodamine WT was injected into well 11S on May 10, 1976, one day prior to injecting tritium into the same well. The tracer was injected directly into the upper portion of the aquifer, immediately below bed N-11. To ensure that all the tritiated water reached the ground water without adhering to the walls of the well casing, well 11S was continuously flushed by pumping into it from well 11D for approximately three weeks. After the flushing period, well 11S was sampled and found to contain no tritium. Monitoring continued daily in all surrounding wells, at the seep areas, and in West Brook through August without recovery of any dye or tritium. There are a number of possible reasons why this first tracer study was unsuccessful. One possible reason could have been that the tritium traveled as a plug and was not detected because the sampling frequency was not short enough. If the tritium did not move into the aquifer as a plug, it could have been small enough to have passed by all monitoring wells without being detected. Another obvious explanation could be that not enough tritium was used to be detected after dilution in the ground water and infiltrating sewage effluent. The most reasonable explanation is that the tritium moved through the aquifer as a narrow tear-like plume and did not pass any of the sample wells and was missed in the seeps either because it became diluted in transit to this station or passed through between sampling periods. To assure that this situation could not occur in subsequent studies, tritium and dye were mixed with sewage effluent in infiltration basins then allowed to percolate into the aquifer covering a much larger area, thereby, greatly increasing the chance of detection at downgradient stations.

The second tracer study began on June 30, 1976, in bed N-4 (Figure 6). The study was conducted in the manner described above to evaluate both the vertical flow and the direction and time of horizontal flow. Flooding of bed N-4 began about 8:00 am and at 11:00 am, while the bed was being flooded, 108 kg (240 lbs) of NaCl and 18 kg (40 lbs) of 20 percent rhodamine WT dye solution were mixed in a 379 liter (100 gal) tank and applied to the bed influent on the splash apron. To ensure uniform distribution of the tritium, the tritiated water (0.1 Ci of tritium) was mixed with city tap water in a 95 liter (25 gal) tank from which it was discharged onto the bed N-4 splash apron beginning at 2:00 pm. Immediately after introduction of tritium in bed N-4, the wells located between bed N-4 and West Brook were sampled daily and analyzed in the North Hall Laboratory for the three tracers. On July 31, 1976, the Tri-Carb liquid scintillation counter became inoperative, and all subsequent samples were counted on the Beckman system in the laboratory located at Lake George. In order to assure retrieval of the tracers, an additional well (well 14) was installed by July 3, 1976, approximately 15 m (50 ft) from the northeast corner of sand bed N-4. This well was downgradient of bed N-4 and located in the top meter (3 ft) of the aquifer where there would be a high probability of intercepting the tracers. Sand bed N-4 was monitored until the tritium no longer appeared in the sewage ponded on the bed. After approximately one week of sampling, during which time the bed was flooded twice, no tritium could be detected. The initial infiltration period for the tracer spiked flooded bed was approximately four days while the second dosing took approximately three days to infiltrate. Sampling was continued on a weekly basis until August 1976 without detecting any tracer at any sampling station including well 1 which was located in bed N-4 at a depth of about 2 m (6 ft) into the aquifer. Sampling ceased in August, and preparation for a third and final tracer study was begun.

In view of the failure to detect any tracers in the first two tritium tracer studies, it was important that the third and final tracer study be successful. In order not to lose the tracer by means of dilution, it was decided the south sand beds should be used to conduct the test since there is less distance to the ground water and less ground water available for dilution. Sand bed S-3 was chosen for the tracer study since this bed had only 5.5 m (18 ft) of sand above the bed rock, and the top of ground water was only about 4.2 m (14 ft) from the sand bed surface (aquifer thickness of about 1.3 m (4 ft). Furthermore, observation well 5 was located in this sand bed and penetrated the aquifer. A ring of additional observation wells was installed adjacent to the south beds on the downgradient side (Figure 49). These new wells were required since all other observation wells were downgradient some distance from bed S-3, and ground water monitoring close to the dosed bed was imperative. During well construction, the intent was to locate the well screens alternatively at the top and bottom of the aquifer. It may be seen from the summary of the data in Table 14 that since the aquifer was less than 2 m (7 ft) thick, it made little difference whether the 0.7 m (7 ft) long well point was located at the top or bottom of the aquifer. A number of attempts to install observation well 21 along the north edge of bed S-4 was conducted and rock was encountered each time; therefore, its installation was abandoned due to lack of time to locate a suitable site. Wells 15 through 19 were installed during August 1976, and well 20 was completed the second week of September. Sampling was begun on these wells on August 29, 1976, to determine the background levels for the three tracers which would be introduced into the ground water during the final tracer test.

The tracers for the final study were introduced into bed S-4 on November 1, 1976, four hours after start of dosing. This bed had been recently cleaned in preparation for the test and had a relatively high infiltration rate as noted by the fact that it took four hours to maintain surface ponding. A factor influencing the four hour filling period was the low flows coming into the treatment plant. The pump which supplied sewage effluent to the south

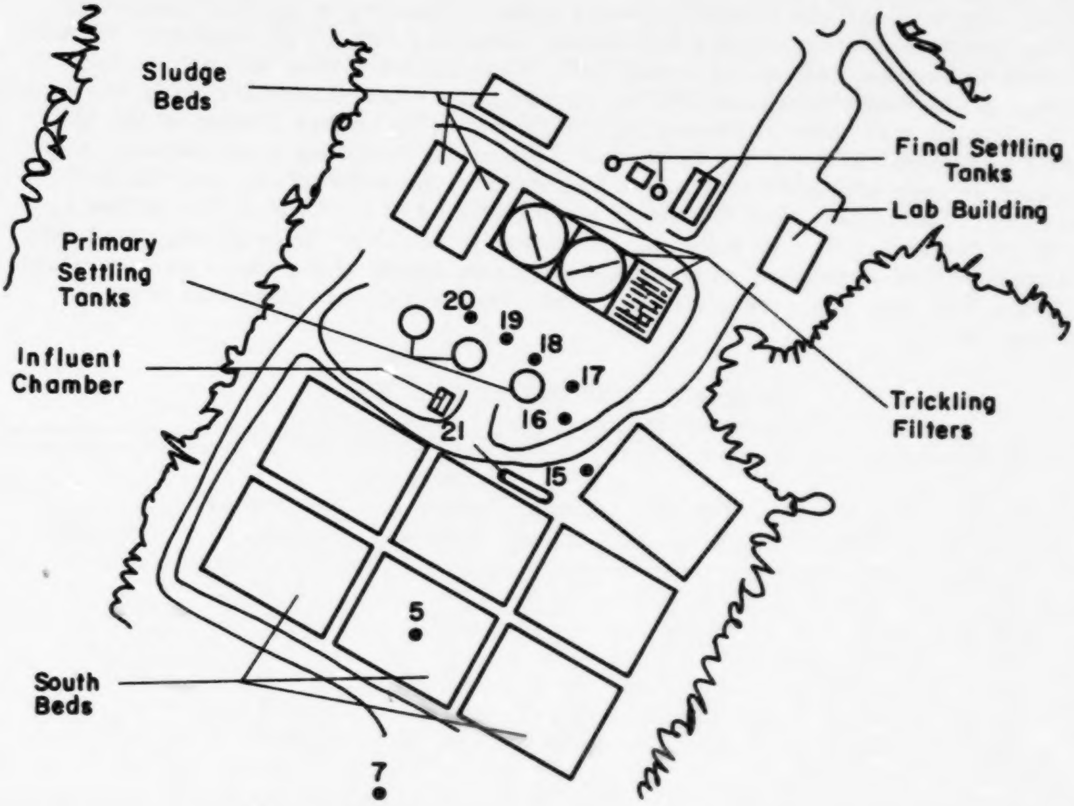

Sludge Beds
Final Settling Tanks
Lab Building
Primary Settling Tanks
Influent Chamber
Trickling Filters
South Beds
20
19
18
17
16
21
15
5
7

Figure 49. Plan Showing Location of New Wells Installed Specifically for the Tracer Studies.

sand beds was operating intermittently due to the low plant inflow. The cycle during the flooding period was 8 minutes of pumping and 8 minutes of resting, which obviously lengthened the normal time of filling. At 12: 40 pm 55 kg (25 lbs) of 20 percent rhodamine WT was added during one 8 minute dosing cycle. The alternating pumping cycle and the wind appeared to uniformly mix the dye in the flooded bed. Starting at 1: 00 pm, 110 kg (50 lbs) of KCL was added in two equal doses over two dosing cycles. Each dose of KCL was first mixed in a 190 liter (50 gal) plastic drum prior to application at the splash apron in the sand bed. Finally, the 0.1 Ci of tritium was diluted in a 60 liter (16 gal) container and added to the applied effluent at the splash apron beginning at 1: 50 pm and continuing through the 8 minute dosing cycle. Flooding of the bed continued for an additional hour and one half before sampling, which allowed time for the tracers to become well mixed in the bed. Samples were then taken from each corner of the bed to evaluate how well mixing had been accomplished in the flooded bed. Dosing was then continued until 8: 00 am on November 2 after which the bed was allowed to dry. By November 3 when the sampling team arrived, the applied sewage effluent and tracers had entered the subsurface, and the bed was dry. Subsequent bed floodings were initiated at 4: 00 pm on November 4, 8: 00 am November 6, and 4: 00 pm November 8. Each of these dosings was for a single period consisting of approximately one fourth of the day's sewage plant inflow. The bed was essentially dry when the next observation was made on November 11.

TABLE 14. DATA FOR SPECIAL WELLS
(Elevations in m. Above Mean Sea Level)

Location	Steel or Plastic	Top of Well	Ground Surface	Approx. Ground Water	Bottom of Point	Bedrock
15	S	148.3	147.8	140	139.9	-
16	S	147.9	147.3	139.6	137.9	137.9
17	S	148.0	147.3	138.3	137.3	-
18	S	148.4	147.9	137.3	135.6	135.6
19	S	148.4	148.0	137.3	136.6	-
20	S	148.4	147.9	137.3	135.8	135.8
21	S	-	-	-	-	-

Sampling indicated the tracers were being retrieved in the observation wells. Both the dye and tritium appeared in well 5 on November 3, 2 days after dosing. As stated earlier, well 5 was located in the aquifer directly below sand bed S-3. The dye and tritium was observed in well 15 on November 8, in wells 16, 17, and 20 on November 9, and well 18 on November 14. After well 19 had been installed, apparently the ground water elevation dropped below the well screen, and samples could not be retrieved. The dye was observed to move farther down-gradient as indicated by its first appearance in wells 9 and 6B on December 4 and 27, respectively.

The patterns of occurrence of the tracers in the observation wells are shown in Figures 50-57. In all of the figures, day O corresponds with the time of application of the tracers into the flooded sand bed S-3 on November 1, 1976. In well 5 (Figure 50), which was located in the center of bed S-3, the tracers appeared on the second day after application with the tritium arriving a few hours prior to the dye. These tracers peaked on the 3rd and 4th day and then dropped sharply to background by the 8th day. In well 15 (Figure 51) the first appearance of the tracer was on the 7th day. A second lower peak occurred in this well on the 10th day possibly indicating pulsing due to the intermittent bed loading. By the 12th day the tracer concentrations returned to their background levels. In well 16 (Figure 52) the first appearance of the tracers was on the 8th day, with the highest concentrations of both the tritium and dye occurring on the 9th day. The tracers indicated a second peak on the 12th day, again indicating pulsing within the aquifer. By the 13th and 14th days, the tracers had returned to background levels. The highest concentration of tritium observed in this study from any sampling station occurred in well 17 as noted in Figure 53. Well 17 is located approximately in the middle of the ring of new observation wells installed for the final tracer study. The first indication of tritium was observed on the 8th day with the highest concentration of both tracers occurring on the 9th day. This peak was followed by a second peak on the 12th day suggesting pulsing characteristics described earlier. By the 14th day the tracers in well 17 had returned to background. Well 18 was located behind the rock which was encountered during attempts to install well 21. Apparently, the flow had to pass around this rock to reach well 18 since the first appearance of the tracers above background levels was not observed until the 13th day as shown on Figure 54. The peaks were much lower and broader than other wells indicating more dilution due to longer contact time in the aquifer. The peaks occurred on the 18th and 19th days followed by a slow return of tracer concentration to background levels. Well 20 was located west of well 18, apparently on the opposite side of the rock previously mentioned. Tracers first appeared in well 20 on the 8th day and peaked about the 11th day, indicating the flow may have passed well 20 prior to reaching well 18. The tracer concentrations slowly returned to background as in well 18. The rhodamine WT was detected in wells 9 and 6B beginning on the 33rd and 56th days, respectively, as shown in Figures 56 and 57. The tritium was not recovered in either of these wells or any other sampling locations although sampling was continued through June 1977. Since rhodamine WT identified as that added to bed S-3 was positively monitored in wells 9 and 6B and no tritium could be detected above background in these wells, it must be deducted that the tritium concentration was diluted to background level before reaching these wells.

Calculations were made of the time of flow to the various observation wells using data collected in the third tracer study as shown in Table 15. Accurate measurements were made of the distance from the north edge of sand bed S-3 to the individual observation wells. The first appearance of the tracer was utilized as the basis for determining the velocity of flow within the ground water. The reason for using the first appearance of the tracer for horizontal velocity calculations was that the tracer was infiltrated over a wide surface area (the bottom of sand bed S-3) resulting in considerable dispersion by the time it reached the saturated

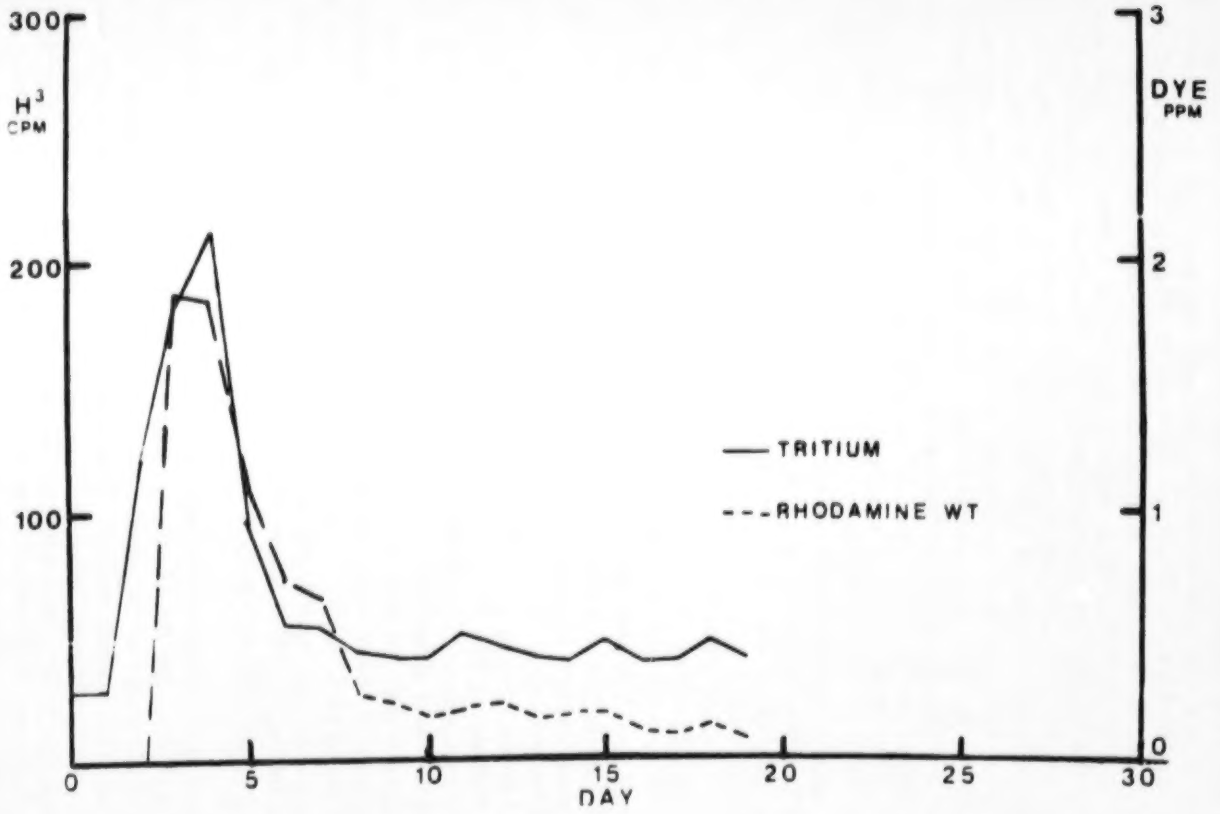


Figure 50. Occurrence of the Tracers in Observation Well 5.

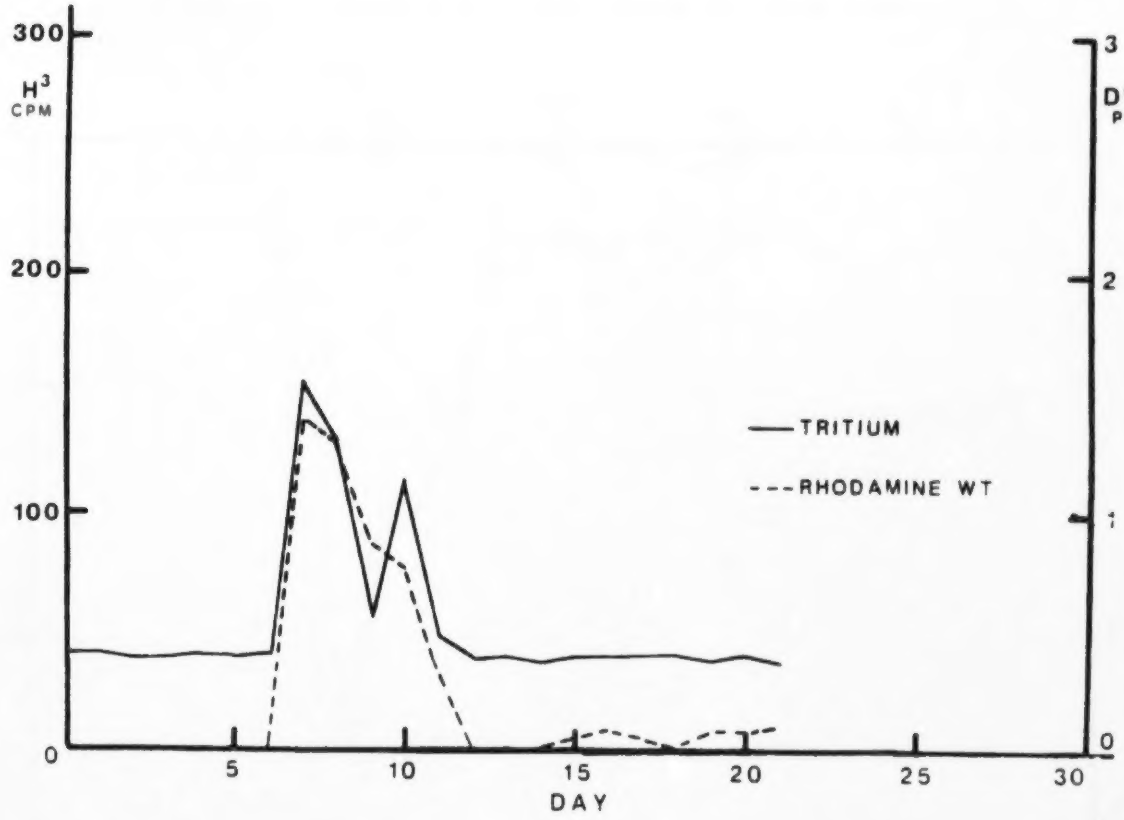


Figure 51. Occurrence of the Tracers in Observation Well 15.

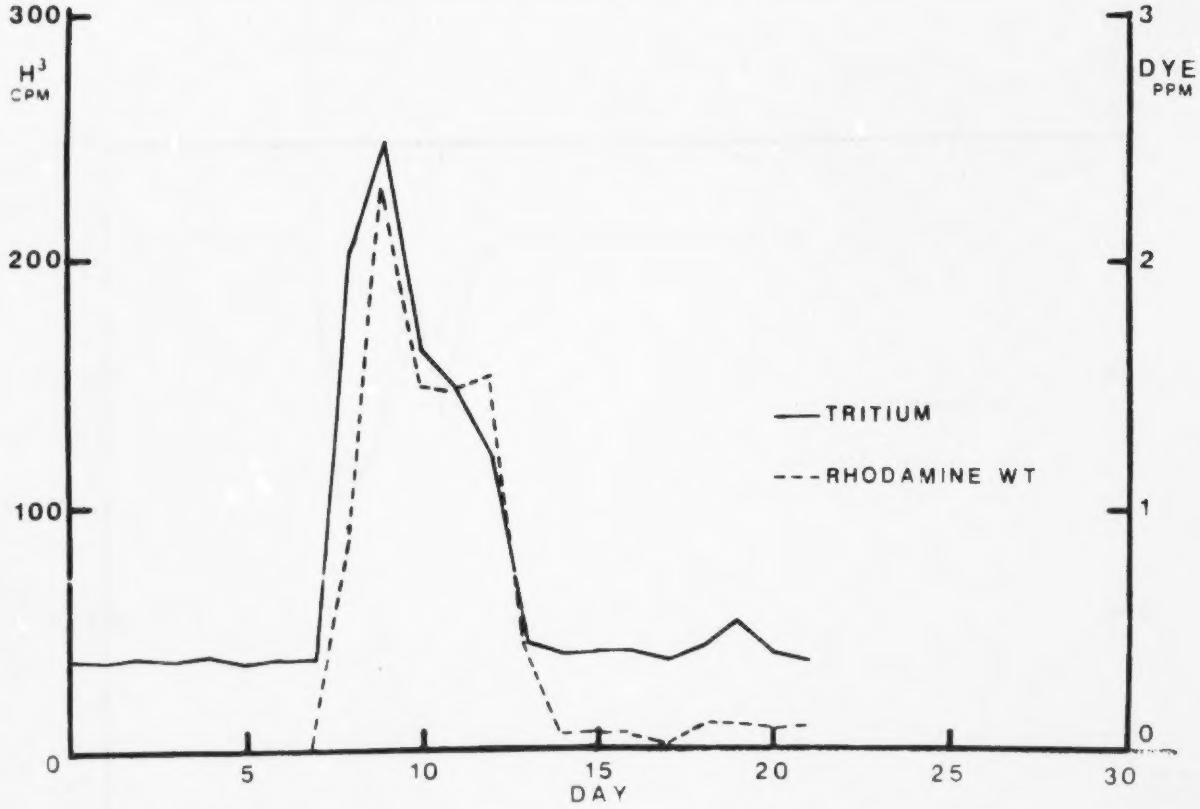


Figure 52. Occurrence of the Tracers in Observation Well 16.

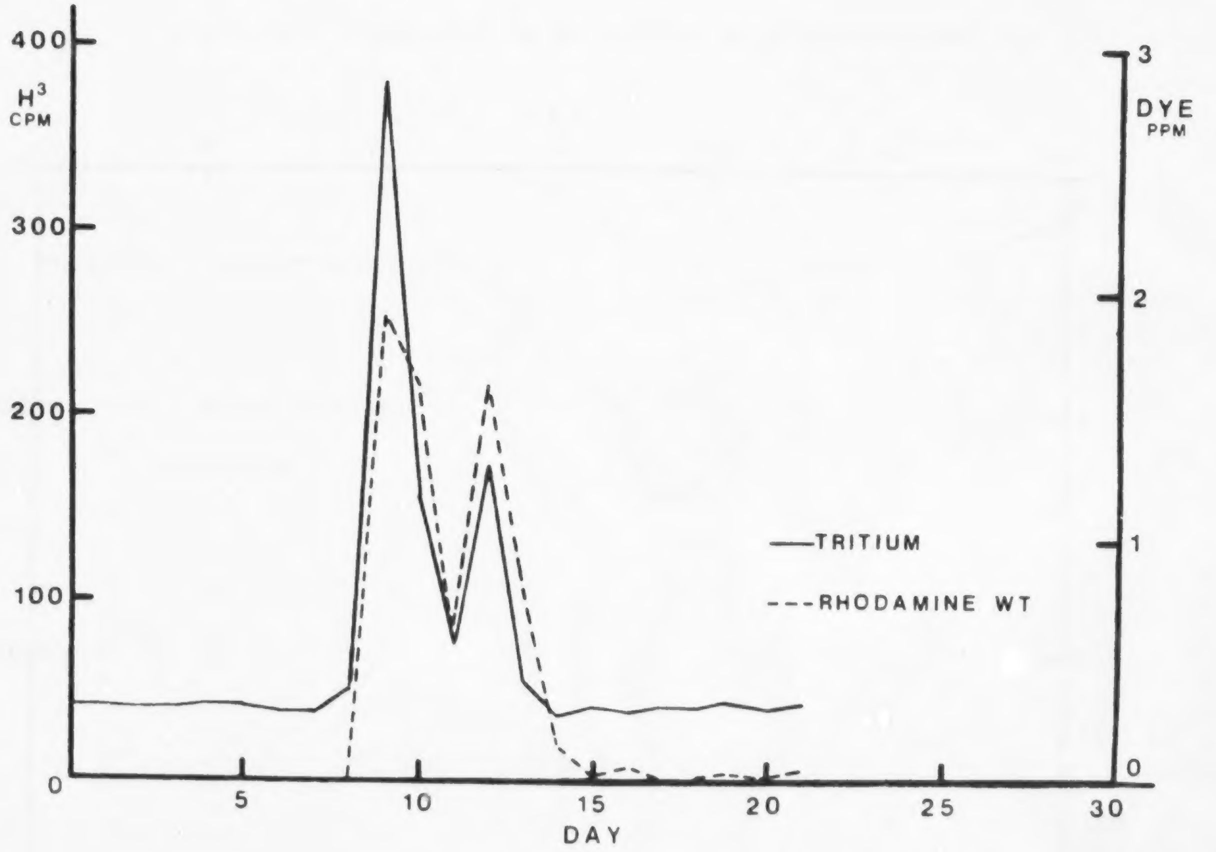


Figure 53. Occurrence of the Tracers in Observation Well 17.

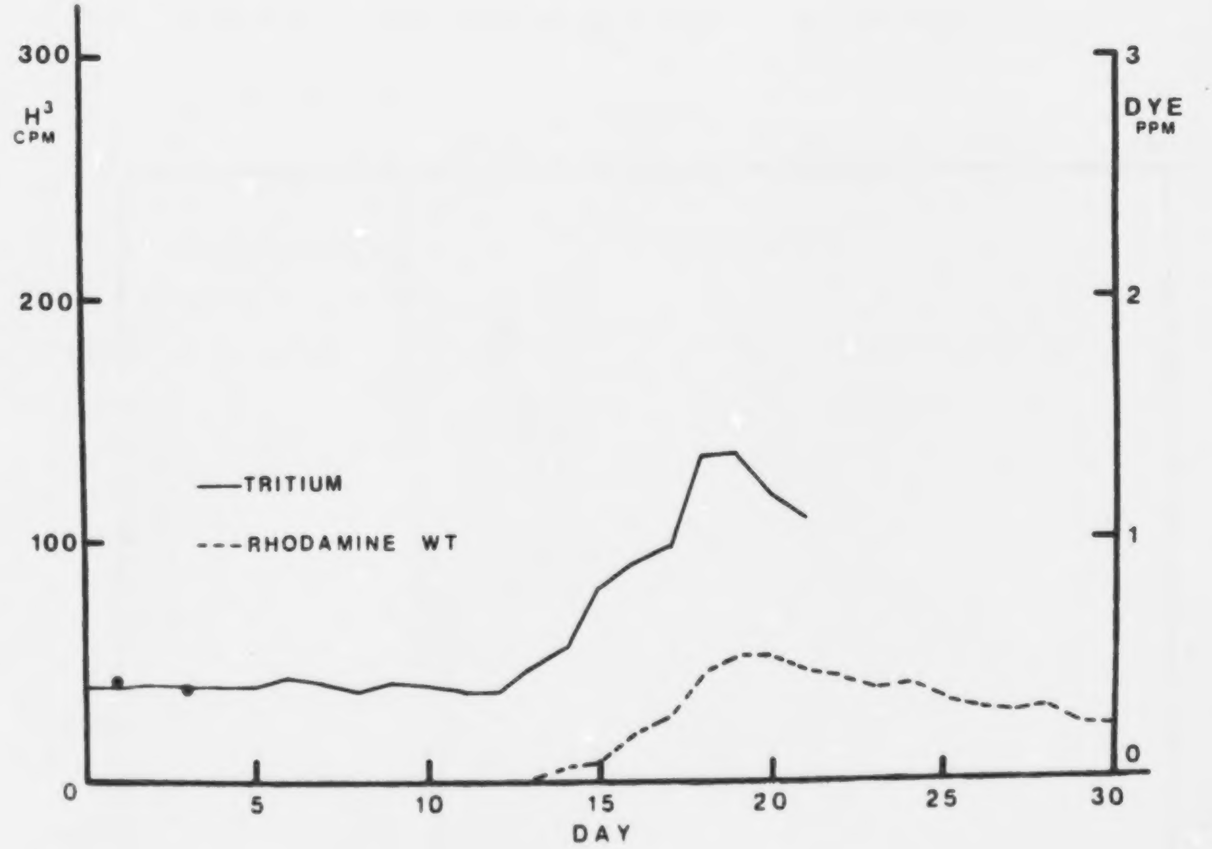


Figure 54. Occurrence of the Tracers in Observation Well 18.

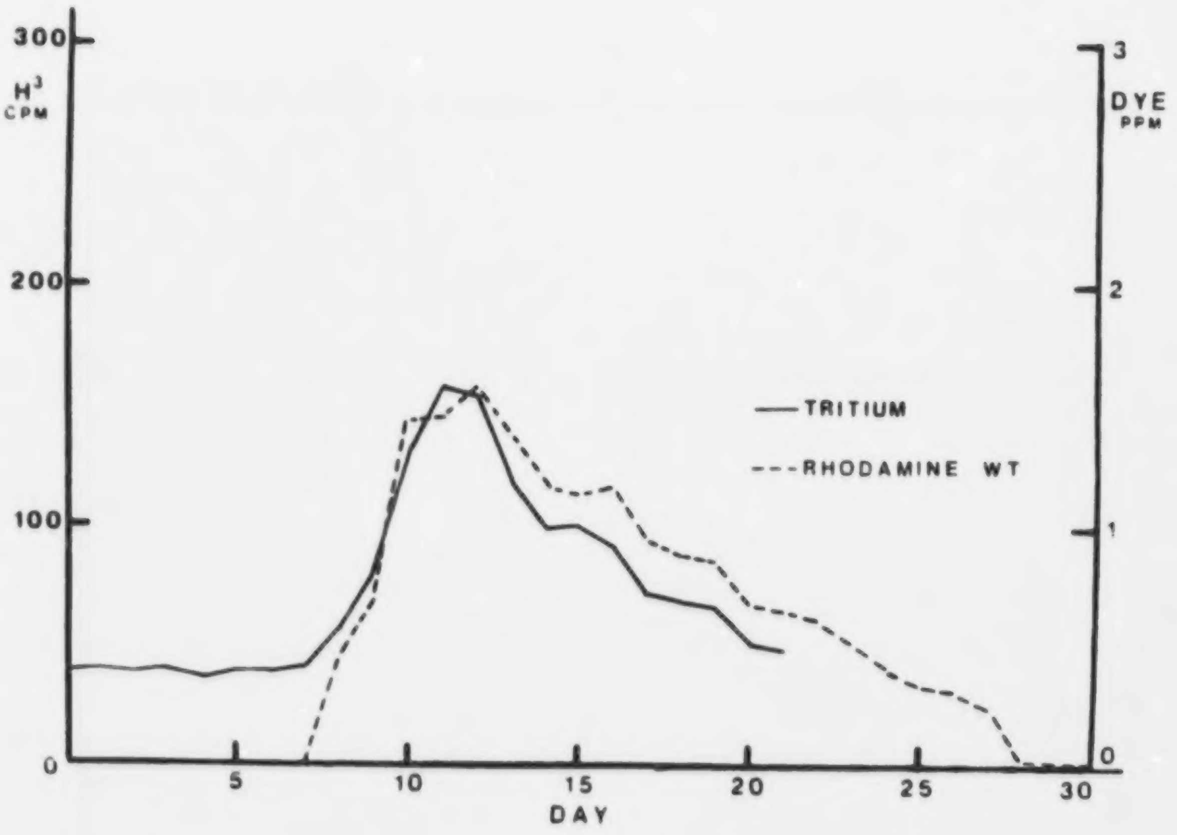


Figure 55. Occurrence of the Tracers in Observation Well 20.

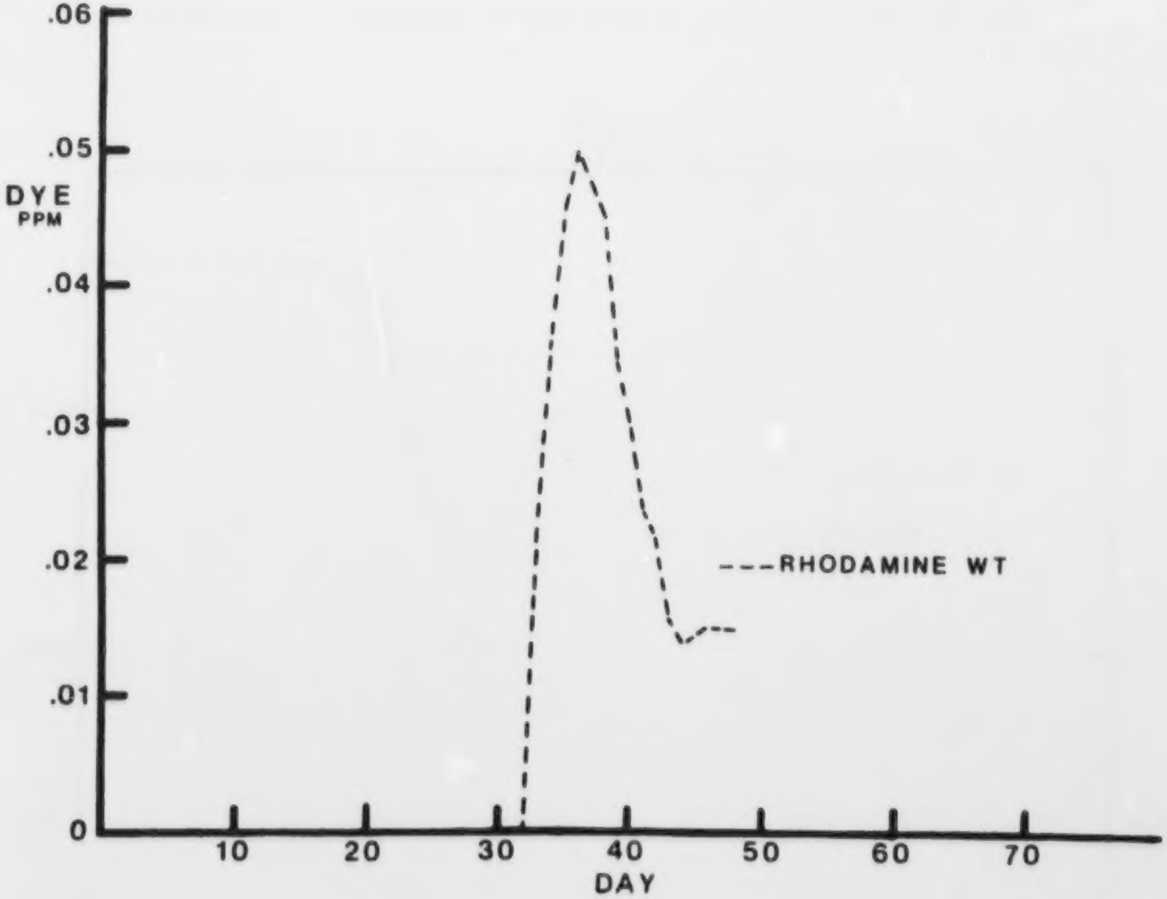


Figure 56. Occurrence of the Rhodamine WT Tracer in Observation Well 9.

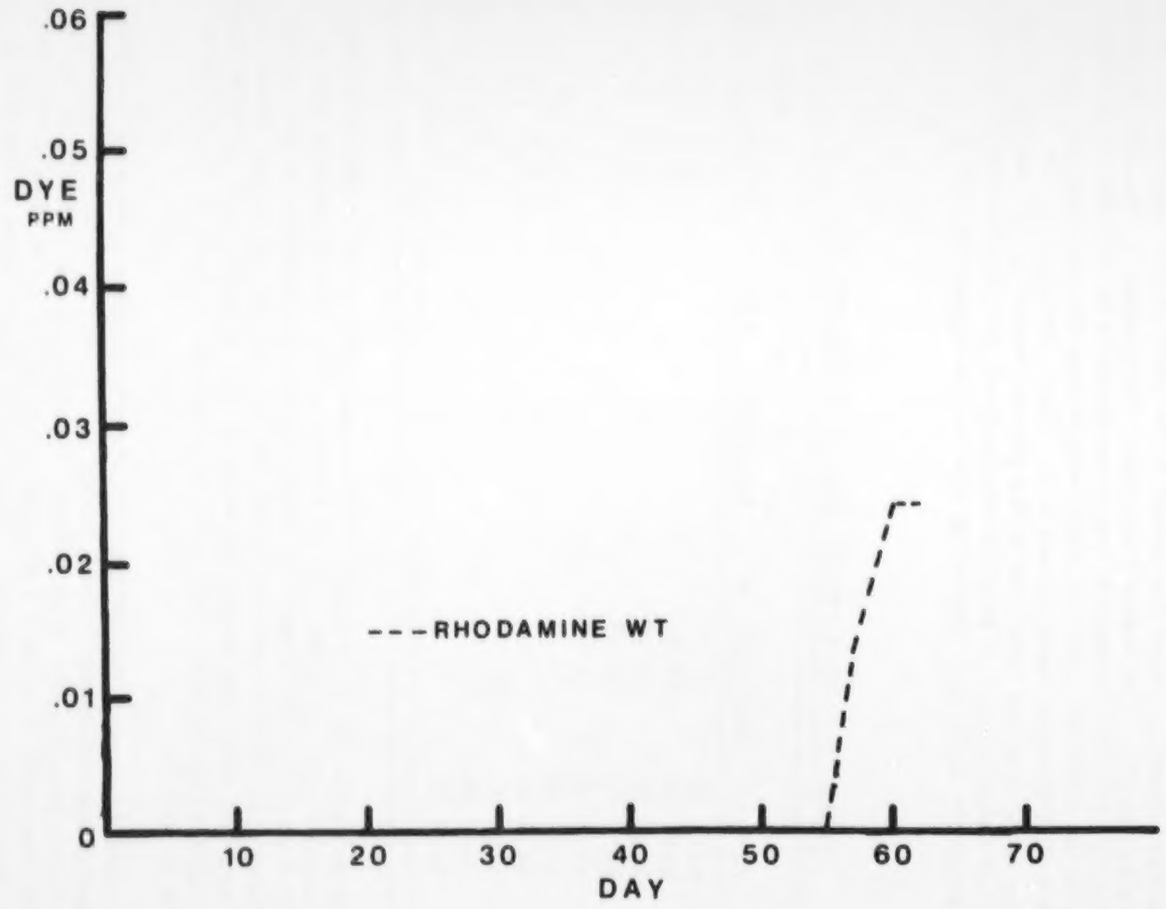


Figure 57. Occurrence of the Rhodamine WT Tracer in Observation Well 6B.

zone. The tracer which entered the ground water at the north edge of bed S-3 would reach the observation wells first, and the peak of the tracer would arrive at the observation wells at approximately the time it took for the flow from the south edge of the bed to reach the observation wells. Therefore, basing the flow velocity on the distance from the north edge of bed S-3 to the observation well and the first observation of the tracer in that well should provide the most valid determination of ground water velocities. Furthermore, the start time for all horizontal velocity measurements was considered to be the time of its first appearance in the saturated ground water which was the time that the tracer first appeared in well 5 directly beneath bed S-3. Thus, the time for the vertical flow from the surface of the sand bed to the ground water is not considered in horizontal flow calculations. It must be emphasized that all the results are considered to have an accuracy of ±1 day, including the time to appear in well 5.

TABLE 15. VELOCITY OF FLOW IN SATURATED ZONE

From	To	Distance, m	Time for First Tracer Appearance, days	Rate m/day	ft/day
Bed S-3	Well 15	50.60	5	10.1	33.2
Bed S-3	Well 16	62.64	6	10.4	34.2
Bed S-3	Well 17	69.72	6	11.6	38.1
Bed S-3	Well 18	75.90	11	6.9	22.6
Bed S-3	Well 19	76.50	-	-	-
Bed S-3	Well 20	72.34	6	12.1	39.6
Bed S-3	Well 9	249.63	31	8.0	26.4
Well 15	Well 9	199.03	26	7.7	25.1
Bed S-3	Well 6B	321.11	54	5.9	19.5
Well 9	Well 6B	71.48	23	3.1	10.2

As seen in Table 15, the velocity in the area north of the south sand beds ranged between 10 and 12 m/day (33-40 ft/day). The slower transit time of 7 m/day (23 ft day) to well 18 may be explained by the presence of the large rock area near well 18 that was discussed earlier. Since the liquid would have to flow around this area, the actual distance of flow was probably greater than the 75.9 m (250 ft) measured. The dye was observed in well 9, 31 days after its observation in well 5.

Based upon 31 days of transit time for the dye to move from well 5 to well 9, the velocity was calculated to be 8 m/day (26 ft/day) which was slightly less than the velocity immediately up gradient. Calculating the velocity on the 26 days transit from well 15 to well 9 gave only a slightly lower rate of flow in this area. The dye occurred at well 6B, 54 days after observation of the dye in well 5. Calculations indicate the ground water velocity between these two observation points averaged about 6 m/day (20 ft/day). By calculating the velocity of flow from well 9 to well 6B for the 23 day transit period between these points, an appreciably lower velocity of 3 m/day (10 ft/day) was indicated. This lower velocity could be attributed to either an actually slower flow rate in this area, to lateral dispersion, or to lateral channelization of the tracer which would effectively increase the distance traveled causing the velocity to appear to be slower.

The next sampling well downgradient in the suspected flow path of the sewage effluent was well 12. Unfortunately, both sample wells at site 12 were inoperable during the period of this study. None of the three tracers added in the third study were ever found in wells 2 and 3, in the seepage above Gage Road, in the seepage below Gage Road, or in West Brook downstream of the seepage areas through the conclusion of the study on June 30, 1977. It is unfortunate that the flow could not be traced beyond well 6; however, the results do agree with Beyer's calculations (19) which indicated that a velocity of approximately 11 m/day (36 ft/day) would be necessary to prevent excessive ground water mounding in the area of the sand beds. One minor discrepancy between the data presented in this study and Beyer's is that in the area between wells 6 and 9, Beyer indicated a horizontal velocity similar to that of the area farther upgradient; whereas, this study indicates an apparent horizontal velocity of approximately 1/3 that of Beyer's.

Comparison of vertical velocity measurements of sand beds S-3 and N-11 revealed some very interesting data. The vertical velocity in bed S-3, based on the appearance of the tracer in well 5, indicated a velocity of 2.5 m/day (8 ft/day) while the vertical velocity measured earlier in bed N-11 was 0.7 m/day (2.32 ft/day). The difference in the two values indicates significant differences in sand hydraulic characteristics. These characteristics can be attributed to natural geologic phenomena that occurred during the formation of the glacial delta in which the sand beds are located. During formation of deltas, soil particles are normally distributed with the very coarse material being deposited near the ice melt area while the progressively finer particles are deposited farther downgradient. Thus, particle size distribution greatly effects both the vertical and horizontal rates of water movement at different locations in the delta.

Based on a conservative velocity of 10 m/day in the saturated zone, it should have taken approximately 60 days for the tracers to travel the 600 m (2,000 ft) to West Brook. As can be seen in Table 15, it actually took 54 days for the tracers to travel from S-3 to well 6B which is approximately half the distance between the south sand beds and West Brook. Since sampling was continued for 8 months or approximately 240 days, it was felt that ample time had elapsed for the tracers to have reached West Brook during the study. Since tracers were never observed beyond well 6, it may be concluded that either an insufficient amount had been added or that the tracers had been adsorbed in the soil system prior to reaching sampling points farther downgradient. It is possible that the rhodamine WT dye was adsorbed by the soil or diluted to background levels; however, it is normally considered that tritium is not adsorbed and, therefore, insufficient tritium may have been added to be detected the entire distance required after dilution in the ground water. Since the rhodamine WT was detected at a greater distance from the application point in this study than tritium, it became a better tracer than the small amount of tritium used.

SOIL ANALYSIS AND WELL TESTS

Numerous samples of the sand were collected for analysis but few extensive analyses were performed due to time limitations. The first set of sand analyses was conducted in 1968 by Glavin and Romero-Rojas (29) as shown in Table 16. These analyses were conducted on sand collected from various depths in bed N-13, and the results indicate the filter media is primarily a fine uniform sand.

TABLE 16. SIEVE ANALYSES - BED #13

Sample Location	Effective Size	Uniformity Coefficient
Test boring hole #3 Approx. 30-33' depth	0.19 m	2.6
Test boring hole #3 Approx. 10' depth	0.135 mm	3.4
Center of bed 13" below surface*	0.25 mm	3.6

*Sample collected and analyzed by New York State Department of Health

Additional analyses were performed by the New York State Department of Environmental Conservation (NYSDEC) on samples taken from the well bores of a number of wells during their construction. It may be seen in Table 17 that soil taken during construction of well 1, located in bed N-4, consisted primarily of fine sand. Well 5, located in bed S-3, was also quite high in fine sand. Well 6 had a nearly equal distribution of fine and coarse sand while well 3 in the seepage area had a mixture containing mostly fine sand with relatively large quantities of gravel and coarse sand. Well 2 located near the seepage area above Gage Road consisted primarily of coarse sand. Comparison of the soils at these five locations illustrates that the soil is not very homogenous or that the sampling technique did not accurately characterize the soil at each location. Permeability measurements conducted on the samples from wells 1, 2, and 6 showed considerable variation. Well 2 had a permeability of 19.5 m/day (64 ft/day) while wells 1 and 6 had 8.5 m/day (28 ft/day) and 4.6 m/day (1.51 ft/day), respectively.

One other set of soil samples was collected on January 9, 1975. These samples were forwarded to the US Army Corps of Engineers Cold Regions Research and Engineering Laboratory, Hanover, New Hampshire. The laboratory then sent the samples to the United States Testing Co., Inc., for extensive analysis. These samples were secured at 0.6 m (2 ft) intervals during the installation of the two pumping wells located in bed N-11. Analyses conducted for organic carbon, total phosphorus, organic phosphorus, exchangeable phosphorus, soluble phosphorus, organic nitrogen, and the cation exchange capacity were all high in the top 3 m (10 ft) of the sand bed, but they gradually decreased with soil depth in relatively low values. Analyses made to determine the level of free iron oxides, pH, soluble salts, and the exchangeable cations (calcium, magnesium, potassium, and sodium) indicated little variation with depth.

Permeabilities measured within the aquifer ranged from 0.4 m/day (1.37 ft/day) to 19.5 m/day (64 ft/day). Based on a mean slope of 0.032 m/m from well 11 to well 2 and using the maximum permeability, the estimated horizontal travel time to well 2 would be 2.67 years. This velocity obviously does not agree with that described earlier in these ground water tracer studies.

TABLE 17. SAND ANALYSIS

In Aquifer	Well 1	Well 2	Well 3	Well 5	Well 6
Effective Size, mm	0.08	.37	0.15	0.12	0.15
Mean Size, mm	0.15	1.40	0.27	0.30	0.38
Uniformity Coeff.	2.13	4.32	2.53	2.83	2.93
"S" Shape Curves	?	Yes	No	Yes	Yes
Gravel %	1	16	20	8	1
Fine Sand %	94	13	62	73	58
Coarse Sand %	6	71	18	19	41
Permeability					
m/day	8.5	19.5	-	-	4.6
ft/day	28	64	-	-	15.1

An attempt was made to determine the permeability of the aquifer below bed N-11 using the pumped wells. Well 11-D was used for the tests conducted during August 11, 13, and 18, 1975. The data were evaluated using three methods: time-drawdown, time-recovery, and residual drawdown (13). Of the three methods, the residual drawdown method is preferred for transmissibility determinations, particularly when an observation well is not available, and the pumped well is used for drawdown measurements (13). These tests resulted in permeability values on the lower end of the range of those previously reported by the NYSDEC. All these tests were taken under conditions of partial penetration of the aquifer and should only be considered approximations.

Permeabilities were measured in the unsaturated soil using the borehole test in well points 11A and 11F at depths of 0.6 m (2 ft) and 3.6 m (12 ft), respectively. The results of these tests showed that the permeabilities in these respective zones were 2.9 m/day (9.5 ft/day) and 2.8 m/day (9.15 ft/day). The range of permeabilities calculated by this method again fell within the range of those calculated by the NYSDEC. A final check on the permeabilities was performed using a sample from a 26.2 m (86 ft) depth from bed N-11. These calculations using Hazen's Formula (13), indicated a permeability of 19.4 m/day (63.8 ft/day). The corresponding permeability estimated by the NYSDEC was 2.8 m/day (9.18 ft/day) as shown in Table 18. In general, the velocities calculated from the permeability studies were much lower than those determined in the tracer studies.

TABLE 18. SAND ANALYSIS - BED 11

Depth, m	20.7	23.7	26.2
Effective Size, mm	0.07	0.09	0.15
Mean Size, mm	0.32	0.48	0.70
Uniformity Coeff.	5.57	6.78	6.0
"S" Shaped Curve	Yes	Yes	Yes
Gravel %	4	8	14
Fine Sand %	45	38	26
Coarse Sand %	36	46	56
Permeability			
m/day	0.4	1.3	2.8
ft/day	1.37	4.27	9.18

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